

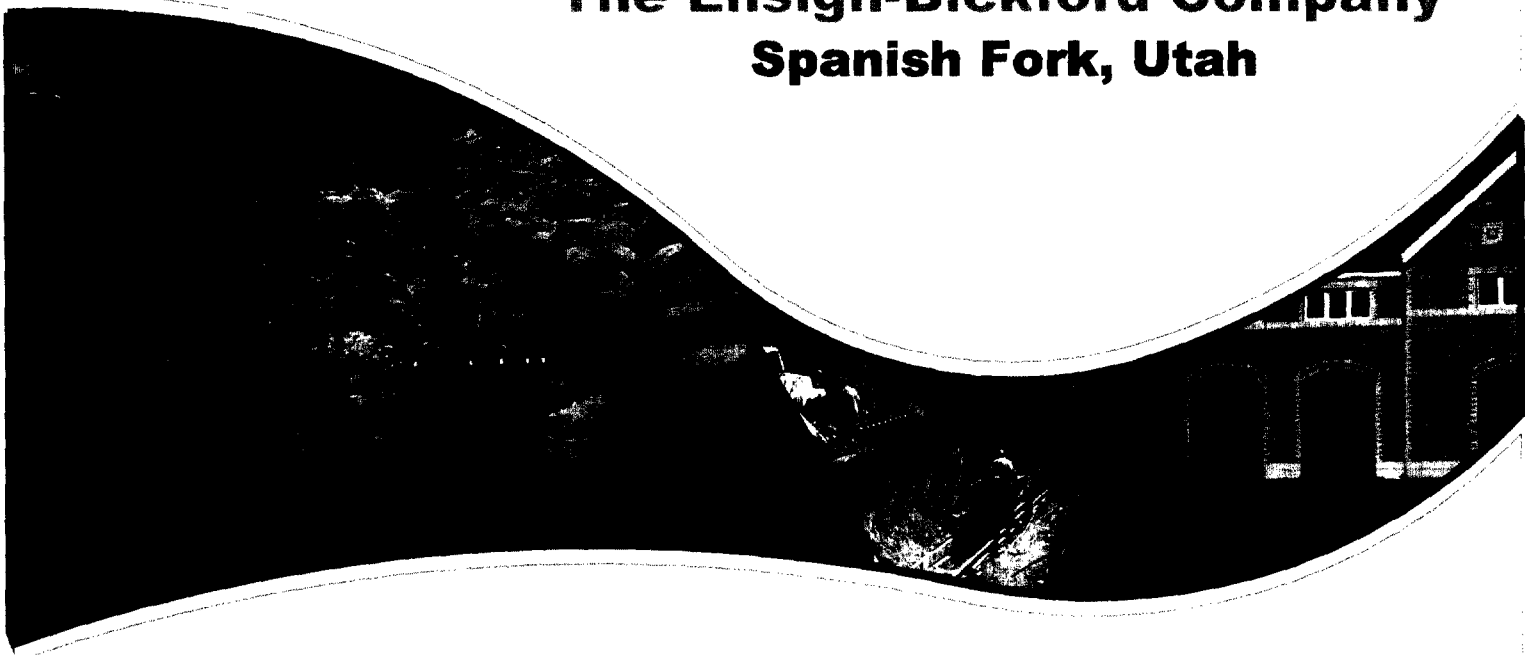
CORRECTIVE ACTION PLAN

RECEIVED

MAY 31 2002

DIVISION OF
WATER QUALITY

**The Ensign-Bickford Company
Spanish Fork, Utah**



CHARTER OAK
Environmental Services, Inc.



Revised May 2002

7.0 CHARACTERIZATION OF CONSTITUENTS OF CONCERN

In accordance with R317-6-6.15.D.1.a.(1-3) of the Utah Administrative Rules for Ground Water Quality Protection (1995), this section provides information regarding the amount, form, concentration, toxicity, fate and transport of solutes.

7.1 Chemical and Physical Properties of CEMs

As noted previously, this CAP addresses ground water impacts potentially resulting from Plant manufacturing activities. Nitrate, measured as nitrate-nitrogen, and eight CEMs have been routinely detected in ground water samples collected from wells open to the regional aquifer within the study area. Two additional CEMs (2,4,6-TNT and 2,6-DNT) have been detected only sporadically and only at low concentrations. The compound 2,4-DNT was added to the parameter list during the first quarter 2001 sampling event. The compounds 2,4,6-TNT, 2,4-DNT and 2,6-DNT were not detected in ground water samples collected during 2001. NG has not been identified above method detection limits in the regional aquifer. Table 7-1 presents a summary of the important chemical and physical properties of the twelve CEMs. Table 7-2 provides descriptions of how these physical and chemical properties may affect environmental fate and transport.

7.2 Environmental Fate and Transport Processes

There are several chemical and physical processes that control the behavior and movement of a chemical species in ground water systems. Several common fate and transport mechanisms are described in the following sections. An in depth review of the theory and mathematical equations used to describe these fate and transport processes is beyond the scope of this report; however, the topics are addressed in sufficient detail so that one can understand how these processes may affect subsurface transport in the study area. The focus of this discussion is on subsurface transport in saturated ground water flow systems. Environmental fate and transport mechanisms for soil, air and biota are discussed and evaluated in the Phase I RFI Work Plan (Montgomery Watson, 1998). Please note that for CACL development purposes, the potential for CEM uptake in selected edible plants was evaluated because these plants could be watered with impacted ground water drawn from privately owned wells (see Appendix E).

7.2.1 Advective-Dispersive Transport

Advective-dispersive transport is the underlying process governing the subsurface migration of solutes. Advective-dispersive transport is a combination of advection, mechanical dispersion and molecular diffusion. Each of these processes is described below.



Table 7-1: List of CEMs and Selected Chemical and Physical Properties
Page 1 of 2

Name	CAS Name	Synonyms	CAS Number	MW (g/mole)	Density (g/cm ³) (25°C)	Vapor Pressure (mmHg)	K _H (atm·m ³ /mole)	Aqueous Solubility (mg/L)	Diffusion Coefficient (cm ² /s)	Log K _{ow}	Log K _{oc}
RDX	Hexahydro-1,3,5-trinitro-1,3,5-triazine	Cyclotri-methylene Tetranitramine; Cyclonite	121-82-4	222 ^a	1.82 ^a	4.03E ⁻⁹ ^{a,c,i}	1.96E ⁻¹¹ ^{a,c,i} (25°C)	28.9 ⁱ (10°C) 42.3 ⁱ (20°C) 60 ^{a,c,i} (25°C) 75.7 ⁱ (30°C)	7.15E ⁻⁶ ⁱ	0.87 ^{a,b,c,i} 0.81 ^{a,c,i} 0.86 ^{a,c,i} 1.1 ⁱ	2.43 ⁱ 2.00 ^{a,c,i} 2.13 ⁱ 1.80 ^b 0.89 ⁱ
HMX	Octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine	Cyclotri-methylene Trinitramine	2697-41-0	296 ^a	1.9 ^a	3.33E ⁻¹⁴ ^{a,c}	2.60E ⁻¹⁵ ^{a,c,i}	1.21 ⁱ (10°C) 2.6 ⁱ (20°C) 5 ^{a,c} (25°C) 5.7 ⁱ (30°C)	6.02E ⁻⁶ ⁱ	0.26 ^{a,c,i} 0.06 ^{a,c,i}	0.54 ^{a,c} 2.83 ⁱ
PETN	Pentaerythritol Tetranitrate	--	78-11-5	316 ^a	1.78 ^c	5.38E ⁻⁹ ^a	1.07E ⁻⁹ ^a	2.1 ^a	5.61E ⁻⁶ ^a	3.71 ^a	3.39 ^{a,h}
TNT	2,4,6-Trinitro-toluene	--	18-96-7	227 ^a	1.65 ^a	5.51E ⁻⁶ ^a (25°C) 1.28E ⁻⁶ ⁱ (20°C) 8.02E ⁻⁶ ^a (25°C)	1.10E ⁻⁸ ^{a,i} (25°C) 1.6E ⁻⁶ ^{ck}	110 ⁱ (10°C) 130 ^j (20°C) 150 ^{a,i} (25°C)	6.71E ⁻⁶ ⁱ (25°C)	2.00 ^a 1.86 ^a	2.72 ^a
2,4-DNT	2,4-Dinitro-toluene	--	121-14-2	182 ^a	1.52 ^a	2.17E ⁻⁴ ^a (20°C) 1.3E ⁻⁴ ⁱ (20°C)	1.86E ⁻⁷ ^a (25°C)	280 ^a (25°C) 270 ⁱ (22°C)	7.31E ⁻⁶ ^a	1.98 ^a	2.40 ^a
2,6-DNT	2,6-Dinitro-toluene	--	606-20-2	182 ^a	1.538 ^a	5.67E ⁻⁴ ^a (25°C) 1.35E ⁻⁴ ⁱ (20°C)	4.86E ⁻⁷ ^a (25°C) 2.17E ⁻⁷ ⁱ	208 ⁱ (25°C) 206 ^a (25°C)	7.31E ⁻⁶ ⁱ	1.89 ^a 2.02 ^a 2.09 ⁱ	1.89 ^{a,h} 1.79 ⁱ 1.28 ⁱ 1.85 ⁱ
NG	Trinitroglycerol	Nitroglycerin	55-63-0	227 ^a	1.59 ^a	1.77E ⁻³ ^a (25°C)	2.71E ⁻⁷ ^a (25°C)	1950 ^a (25°C)	6.95E ⁻⁶ ^a	1.62 ^a 1.77 ^a 2.81 ^a	2.77 ^a 1.66 ^a
EGDN	Ethylene Glycol Dinitrate	Nitroglycol, Glycol Dinitrate	628-96-6	152 ^a	1.49 ^{a,f}	0.0706 ^a	2.52E ⁻⁶ ^a	5600 ^a (25°C) 6800 ^f	8.72E ⁻⁶ ^a	1.16 ^a 2.11 ^a	1.28 ^{a,h}
DEGDN	Diethylene Glycol Dinitrate	Dinitroglycol	693-21-0	196 ^a	1.38 ^{a,f}	0.00593 ^a 0.0036 ^f 0.130 ^f (60°C)	3.83E ⁻⁷ ^a	4000 ^{a,f}	7.05E ⁻⁶ ^a	0.98 ^a 1.13 ^a	2.03 ^a
TEGDN	Triethylene Glycol Dinitrate	--	111-22-8	240 ^f	1.33 ^f	<0.001 ^f (25°C) 1.9E ⁻⁴ ^m (30°C)	9.1E ⁻⁹ ^k	5500 ^f (25°C) 6600 ⁿ (25°C)	--	0.62 ^g 0.62 ^o	0.84 ^h 1.27 ^h
TMETN	Trimethylol-ethaneTrinitrate	Metriol Trinitrate, 2-Methylnitrate	3032-55-1	255 ^d	1.47 ^f	<2E ⁻⁴ ^f (25°C) 1.8E ⁻⁴ ^m (26.5°C)	1.2E ⁻⁷ ^k	150 ^f (25°C) 516 ⁿ (19°C)	--	2.46 ^g 2.00 ^o	2.36 ^h 1.98 ^o
BTTN	Butanetriol Trinitrate	1,2,4-Butane-triol Trinitrate	6659-60-5	241 ^d	1.52 ^f	0.12 ^f (50°C) 0.20 ^f (60°C) 0.028 ^f (25°C)	1.1E ⁻⁵ ^k	800 ^f	--	2.00 ^g 1.90 ^o	1.98 ^h 1.89 ^o

Table 7-1: List of CEMs and Selected Chemical and Physical Properties

Page 2 of 2

Temperature of measurement is 20°C unless otherwise indicated.

-- Indicates that information not available at this time.

The primary reference document may not be identified. The primary reference is identified in the cited document.

Information Sources:

^a Organic Explosives and Related Compounds: Environmental and Health Considerations, Burrows et. al., March 1989

^b Toxicological Profile for RDX, USDHHS, Atlanta, GA, June 1995

^c Treatment of RDX and HMX Contamination, Amarillo National Resource Center for Plutonium, March 1998

^d www.chemfinder.com

^e Owens Compliance Services, Inc., MSDS

^f Trojan Corporation Product Information Sheet

^g Calculated using methodology described by Meylan and Howard (1995). K_{ow} values estimated based on chemical structure using SRC online KowWin program.

^h Log K_{oc} values estimated following Lymen and Loreti (1987). $\text{Log } K_{oc} = 0.824 \text{ log } K_{ow} + 0.328$

ⁱ Review of Formulations for Processes Affecting Subsurface Transport of Explosives (McGrath, 1995)

^j Toxicity and Metabolism of Explosives (Yinon, 1990)

^k Calculated from vapor pressure and solubility

^l Kenega, 1980 (as cited in HSDB, 2001)

^m Holleman et. al. (1983)

ⁿ Syracuse Research Corporation (SRC), 2001. <http://esc.syrres.com/efdb/datlog.html>

^o Chou and Jurs (1979). Program called CLOGP available online at www.biobyte.com/bb/prod/clogp40.html

TABLE 7-2: DESCRIPTION OF PHYSICAL AND CHEMICAL PROPERTIES OF ORGANIC CONSTITUENTS THAT AFFECT ENVIRONMENTAL FATE AND TRANSPORT

Molecular Weight	The molecular weight of a compound generally affects its mobility. As the molecular weight of compounds with homologous chemical structure increases, the tendency for a compound to be adsorbed to soil increases and solubility in water decreases.
Density	The mass of a chemical per unit volume. Density is used to estimate whether a contaminant or mixture of contaminants will float on the ground-water surface (LNAPLs with a density less than 1 g/cm ³) or sink (DNAPLs with a density greater than 1 g/cm ³).
Vapor Pressure	The concentration of a chemical in the vapor state that is in equilibrium with the pure chemical liquid or solid. Vapor pressure provides a qualitative measure of the transfer of a compound from the liquid to gaseous phase in unsaturated environments or surface releases.
Henry's Law Constant	Henry's Law Constant (K_H) is a measure of chemical partitioning between air and water at equilibrium. It applies to the liquid-air contact in unsaturated environments or the liquid-air contact at the ground surface. The higher the Henry's Law Constant, the more likely a chemical is to volatilize than remain dissolved in water.
Aqueous Solubility	Quantifies the maximum concentration of a compound that will dissolve in an aqueous solution at equilibrium and at a specified temperature. The aqueous solubility of CEMs has a direct relationship with temperature. CEM solubility will decrease as temperature decreases.
Aqueous Diffusion Coefficient (D_w)	Expresses the rate at which a compound will migrate in water from regions of higher concentration to regions of lower concentration through molecular diffusion processes alone. The diffusion coefficient generally is directly proportional to temperature. The effects of diffusion are generally considered insignificant in certain advection-dominated systems; however, diffusion may play an important role in solute retardation in heterogeneous environments having abundant fine-grained strata.
Octanol-Water Partition Coefficient (K_{ow})	The octanol-water partition coefficient (K_{ow}) is a standard indicator of how a chemical partitions between the aqueous phase and an organic phase, with octanol serving as a surrogate for fat and other biological tissues, soil organic carbon or other organic immiscible liquids. K_{ow} is defined as the ratio of a dissolved compound's concentration in octanol to its concentration in water when the two liquid phases are in equilibrium. For environmental purposes, K_{ow} is not employed in its own right, but is frequently used to predict whether an organic contaminant will tend to solubilize in a non-aqueous phase liquid, adsorb to the organic constituents of soil, or accumulate/concentrate in the tissue of fish, birds, vegetation, or other biota.
Organic Carbon Partition Coefficient (K_{oc})	The organic carbon partition coefficient K_{oc} is defined as the ratio of the concentration of adsorbed chemical per g organic carbon to the concentration of a chemical per mL of solution. K_{oc} values can be measured directly or estimated from K_{ow} . When K_{oc} is normalized for organic carbon content of the soil (f_{oc}) the resulting distribution coefficient (K_d) provides a measure of the extent that a compound will adsorb to soil or will remain solubilized in water. As the K_{oc} increases, adsorption to soil increases. In general, a compound that has a K_{oc} of 50 or less is considered very mobile and will remain solubilized in water (Dragun, 1988).

7.2.1.1 Advection

Advection is the passive movement of a solute with flowing ground water at the average linear velocity. The amount of solute being transported is a function of the solute concentration in ground water and the quantity of ground water flow. Theoretically, in the absence of other transport and fate mechanisms, a body of solutes being acted on solely by advection would maintain the same concentration as it moved from point A to point B. Solute transport by advection alone results in a sharp concentration front at the leading edge of the solute mass as the invading solute replaces the all pore fluid in the porous media. This is known as plug flow. Advection is the most significant transport mechanism acting in the study area and is largely responsible for the overall distribution of solutes. In the environment, mechanical and chemical forces disburse the solute front.

7.2.1.2 Mechanical Dispersion

Ground water moves at rates that are both higher and lower than the average linear velocity. According to Fetter (1993), there are three basic reasons for this phenomenon: (1) As water moves through the pores, it will move faster in the middle of the pores than along the edges of the pores; (2) Some of the fluid particles will travel along longer flow paths in the porous media than other fluid particles to travel the same linear distance; and, (3) Pores vary in size and fluid will flow through larger pores at a higher velocity than in smaller pores. As noted previously, if impacted ground water were to travel at a uniform velocity, it would displace non-impacted ground water and create a sharp concentration front at the interface between the invading and receiving ground waters. Because all ground water does not travel at the same rate, mixing occurs along the flow path. This mixing is called mechanical dispersion and results in the reduction of solute concentrations at the advancing solute front. Advection must be present in order for mechanical dispersion to occur. Mechanical dispersion is independent of the chemical properties of the solute.

Mechanical dispersion is a three-dimensional process and causes the spreading of a solute front in directions longitudinal and transverse to the direction of ground water flow. Typically, dispersion along the prominent flow direction, longitudinal dispersivity, is much greater than that of dispersion perpendicular to the main flow direction or transverse dispersivity. Although actual dispersivity ratios are highly site specific, in sedimentary aquifers vertical transverse dispersivity is typically less than ten percent of the longitudinal dispersivity and lateral transverse dispersivity is generally on the order of ten to thirty percent of the longitudinal dispersivity (McGrath, 1995). Mechanical dispersion is also scale dependent and is observed to increase asymptotically as a function of distance from the source. As the solute spreads, it encounters different lithologies, which affect the cumulative dispersion. Field scale dispersion is often referred to a macrodispersion. Because of the highly heterogeneous nature of deposits in the study area, mechanical dispersion is probably a significant transport factor.



7.2.1.3 Molecular Diffusion

Molecular diffusion is a physiochemical process resulting from the random motion of molecules, resulting in a net migration down a chemical gradient. In the simplest sense this is the movement of solutes from areas of high concentration to areas of low concentration. The rate of molecular diffusion is a function of the chemical properties of the solute; the diffusion coefficient, an empirical constant, determines the rate of diffusion for a particular compound. In most hydrogeologic scenarios the spreading of a solute front by molecular diffusion is considered negligible relative to mechanical dispersion and is generally ignored or lumped into the hydrodynamic dispersion coefficient. McGrath (1995) reports that some researchers have suggested that diffusion in soils becomes relatively insignificant as seepage velocities exceed approximately 5.58 ft/day. This velocity is relatively high and within the range of ground water flow velocities expected within some portions of the regional aquifer. Therefore, diffusion is probably a noteworthy transport mechanism under many ground water flow regimes. Because of the heterogeneous nature of materials throughout the study area, diffusion may play an important role in solute migration. There are abundant saturated fine-grained silt- and clay-rich lenses and layers that are known to, or are likely to contain impacted ground water in the study area. Chemical migration by advection and mechanical dispersion through these low permeability units will be extremely slow. Diffusion may play a part in the migration of solutes in ground water from more permeable deposits into adjacent low permeability deposits. Conversely, it is possible that as ground water in more permeable units becomes cleaner through active restoration or natural attenuation, concentration gradients may form between ground water in pore spaces within the more and less-permeable units. These concentration gradients may result in the diffusion of chemicals from the less permeable deposits into the ground water flowing through the more permeable deposits, resulting in longer term, low concentration impacts to ground water.

Aqueous diffusion coefficients have been calculated for several CEMs and are summarized in Table 7-1. Based on available information, aqueous diffusion coefficients for CEMs range from approximately 5.6×10^{-6} to 8.7×10^{-6} cm²/s.

7.2.2 Dilution

According to Wiedemeier et. al. (1996), dilution results in the reduction of solute concentrations and an apparent decrease in the total mass of solutes in a system due to the introduction of additional water to the system. Dilution does not destroy or transform constituents. For the purposes of this CAP, dilution means a reduction of solute concentrations caused by the addition of clean ground water recharge to an aquifer; in this case, the regional aquifer. Dilution probably plays an important role in decreasing chemical concentrations in the study area, particularly in areas of high recharge. This may explain, at least in part, the decrease in concentrations observed in wells screening the Crowd Canyon alluvial fan deposits. The declining concentration trends generally coincide with a substantial increase in water levels in this area that is related to higher



than normal levels of precipitation from 1994 – 1998. Dilution may also play a role at the extreme northern and southern edges of the study area where a substantial volume of clean water recharges the ground water system from Hobbie Creek and the Spanish Fork River.

7.2.3 Sorption

For the purposes of this discussion, sorption refers to the interaction between a dissolved aqueous phase solute and the sediment particles that comprise the aquifer matrix. Sorption processes encompass both adsorption and desorption processes.

According to McGrath (1995) adsorption is the dynamic process by which dissolved solutes accumulate at an interface, in this case the sediment particles that comprise the aquifer matrix. The adsorption process is generally reversible and the reverse process is called desorption. At equilibrium, the local rates of adsorption and desorption are balanced. Irreversible sorption implies the formation of a stable, covalent bonds between the solute and solid. McGrath considers irreversible sorption to be an abiotic reaction.

Sorption processes can be complex and are not well understood for CEMs. The sorption of CEMs is the topic of ongoing research in the scientific community. Sorption is dependent on the physiochemical properties of both the solid and the solute and their response to complex and variable environmental conditions. In a heterogeneous environment with multiple solutes it is probable that multiple sorption mechanisms are active. Because of these inherent complexities, environmental practitioners generally use what is called the “local equilibrium assumption” (LEA). The LEA presumes ground water flow is sufficiently slow relative to reaction kinetics (adsorption, dissolution/precipitation, etc.) so that local process equilibrium is considered instantaneous. Use of the LEA is often necessary because sufficient data is often not available to consider the more complex alternatives. The simplest sorption models assume ideal behavior, which requires local equilibrium, a linear relationship between sorbed concentration and aqueous concentration, and a completely reversible sorption reaction. McGrath (1995) points out that for selected CEMs, there is evidence of non-ideal behavior in both laboratory and field studies. McGrath reports four potential mechanisms of non-ideal behavior:

1. Sorption isotherm nonlinearity
2. Sorption-desorption nonsingularity (hysteresis)
3. Rate limited or kinetic sorption reactions
4. Diffusive mass transfer into immobile water regions

Nonequilibrium transport behavior is attributable to either chemical or physical processes, or a combination thereof. Chemical nonequilibrium may be important when sorption kinetics are slow relative to advective-dispersive transport. Physical nonequilibrium results from the slow diffusion of solutes between relatively high advective regimes and low advective regimes. The affects that nonequilibrium sorption processes have on



solute transport are not well understood, particularly at the field scale. While equilibrium conditions may be approached locally after extended contact between the contaminant and the aquifer matrix and assuming that mass flux is near steady state, the LEA may fail where conditions change rapidly such as at advancing solute fronts and in highly heterogeneous materials. According to McGrath, the most significant problem identified in using the LEA is that it may result in the underestimation of clean-up times for pump and treat remediation systems because it ignores rate-limited, mass transfer processes.

Notwithstanding the forgoing, the LEA is applied to this setting in large part because sufficient data is not available to enable a more detailed evaluation of sorption. However, this simplified analysis is sufficient to compare the relative sorptive characteristics of the CEMs that are present in study area ground water and how these sorptive characteristics may affect the distribution of solutes and the remedial action. Sorption processes in the regional aquifer system are likely to differ from that predicted by the simplifying assumptions applied herein.

Assuming equilibrium conditions and ideal behavior, a simple linear isotherm can be used to predict the sorbate concentration from the aqueous solute concentrations.

$$S = K_d C$$

Where:

S = Sorbate concentration ($\mu\text{g/kg}$)
K_d = Distribution coefficient (cm^3/g)
C = Aqueous concentration ($\mu\text{g/L}$)

The distribution coefficient (K_d) is used to characterize the potential for sorption of a constituent in a given hydrogeologic setting. The K_d is a function of the chemical properties of both the solute and the aquifer matrix material. K_d is defined as the ratio of the mass of solute on the solid phase per unit mass of solid phase to the concentration of solute in solution. Ideally K_d values are determined on a site-specific basis, as the chemical properties of the aquifer matrix will vary from site to site and within a site. The distribution coefficient can be empirically measured by simultaneous analysis of soil and water samples collected from a single point in the aquifer. Site-specific K_d values have not been measured for any chemicals present in study area ground water. An attempt was made to calculate K_d values for RDX from field measurements during the construction of the R-1, R-2 and R-3 recovery wells. However, it was determined that the aqueous concentration of RDX collected from borehole samples were not representative of true subsurface conditions so site-specific K_d values could not be calculated. K_d values for chemical compounds can be estimated by multiplying the organic carbon partition coefficient (K_{oc}) by the fraction of organic carbon (f_{oc}) present in the subsurface materials. K_{oc} values have been derived experimentally for several of the CEMs or have been calculated from the octanol/water partition coefficient (K_{ow}). When experimentally derived K_{oc} estimates are not available, a mathematical relationship developed by Lyman and Loretto (1987) has been used to estimate K_{oc} values from K_{ow} estimates available for



certain CEMs. Log K_{ow} and Log K_{oc} values for the CEMs are provided in Table 7-1. According to LaGrega (1994), typical f_{oc} values for unconsolidated aquifer materials vary over an order of magnitude from approximately 0.002 to 0.03, with sand and gravels at the low end of the range and silts and clays at the high end. Site-specific total organic carbon data for the regional aquifer are limited to one sample collected from fine-grained aquifer materials present at the location of R-3 (Owens Western, 1997). The f_{oc} value calculated from these data is 0.009 for the aquifer materials present at the location of R-3 and at the depth from which the sample was collected. Because of the heterogeneity of deposits in the regional aquifer and the lack of site-specific data, it is not appropriate to calculate estimated K_d values for the regional aquifer. Nevertheless, examination of the Log K_{oc} values presented in Table 7-1 allows relative comparison of the sorption characteristics of the CEMs. For a given f_{oc} , compounds having a higher Log K_{oc} will have a higher K_d and therefore stronger sorption characteristics.

Of potential significance for the study area is the potential for solutes, in particular CEMs, that have adsorbed to the aquifer matrix to act as continuing sources of solutes to ground water. Since finer grained units present in the regional aquifer generally have a high clay component it is likely that the K_d values for many chemical constituents would be higher due to the increased organic carbon associated with clay. Based on the foregoing, the concentration of CEMs that may adsorb to the aquifer matrix could be estimated using the linear sorption isotherm. It is not known if a simple linear isotherm is suitable to approximate the sorption of all the CEMs. Also, the linear sorption isotherm provides no limit to the amount of a solute that can adsorb to a solid. This is not the case in the natural environment as there must be an upper limit to the amount of a solute that can be sorbed (Fetter, 1993). As pointed out previously, it is also not known if the sorption of CEMs is completely reversible. Therefore, the amount of sorbed solute may not completely desorb back into the ground water or the rate of desorption may differ. In the absence of site-specific data, the Log K_{oc} estimates presented in Table 7-1 can be used to assess the relative importance of sorption for the various CEMs for a given f_{oc} estimate.

Adsorption also affects the transport of constituents by retarding the rate of solute migration. The solute front of a compound with low K_d value (unlikely to sorb) will pass the reference point before the solute front of a compound with a high K_d . Assuming a simple linear sorption isotherm, one common application of K_d estimates is to calculate the retardation factor (R_f), which is an estimation of the relative rate of chemical migration in flowing ground water. The retardation factor is calculated as follows:

$$R_f = 1 + \rho K_d / n$$

Where:

- R_f = Retardation factor (unitless)
- ρ = Soil bulk density (g/cm^3)
- K_d = Soil/water partition coefficient (cm^3/g)
- n = porosity (unitless)



Assuming a given set of f_{oc} and porosity values, the Log K_{oc} values provided in Table 7-1 provide insight into the potential for retardation of the various CEMs relative to one another.

7.2.4 Biodegradation

7.2.4.1 Literature Reviewed

More than 100 hundred technical papers or other references were reviewed on topics relevant to the discussion in this section. That material has been compiled over a long period of time in the course of conducting this project. A complete reference list of literature reviewed is not provided, but can be provided if requested. However, within the text certain references are included to support statements made where it was believed that they might be of particular interest.

7.2.4.2 Affected Media

For the purposes of this CAP, the affected media is considered to be saturated, subsurface deposits (i.e. the ground water environment). Surface waters are not considered since they are not a part of the constituent migration pathway. Likewise, shallow or surface soils are not considered in this CAP other than as a potential continuing source of impacts to the regional aquifer. The RFI investigation is currently assessing the potential for on-site soils to represent a continuing threat to ground water. Therefore, biodegradation processes considered as relevant or informative must be consistent with the ground water environment found at this site.

7.2.4.3 Mechanisms for Degradation

There are four potential mechanisms for degradation: aerobic biological; anaerobic biological; photolytic; and abiotic. The literature generally supports the simplification that the combined degradation processes and pathways represented by the aerobic and anaerobic conditions are sufficient to describe the degradation process under abiotic conditions. That is to say that abiotic processes do not offer different pathways or breakdown products, but rather introduce an alternative means for stimulating breakdown. As a result of this observation, the specific discussion of breakdown of the parent compounds is focused on aerobic and anaerobic biological processes. As discussed in Section 5.2.13, photolytic processes are not relevant to the ground water environment.



7.2.4.4 *Practical Relevance to Provisional COCs*

Provisional COCs are not confirmed as present in the regional aquifer at this time. Some of these compounds have never been detected in ground water samples from the regional aquifer and are only listed as provisional COCs due to the potential that they may be detected in the regional aquifer in the course of the RFI investigation. Constituents, such as TNT and related DNT compounds have only been detected sporadically and at very low levels, in an area where declining trends have been observed for other constituents. Even if these provisional COCs are detected in future monitoring, they are expected to be at levels close to their detection limits. Additionally, monitoring data to date suggest that these compounds would only be detected in monitoring wells near the source area (facility) if at all. Therefore, these compounds would be located in an area where natural attenuation through dilution is observed as a mechanism in reducing constituent concentrations. The effect of these issues discussed above is to eliminate any practical relevance to the consideration of biodegradation products of these parent compounds. Even if the parent compounds were found to be present, they would be at such low concentrations and detected on such a sporadic basis that potential biodegradation products could not be detected, even if they were present. Therefore, biodegradation products of the provisional COCs are not discussed further in this CAP.

7.2.4.5 *Biodegradation as a factor in Natural Attenuation*

Biodegradation is one component of natural attenuation. In the case of this site, biodegradation is considered to be a minor potential contributor to natural attenuation, especially in comparison to dilution. Evidence of dilution is observed, for instance, in the obvious declining trends of nitrates in certain regional aquifer monitoring wells, especially close to the facility. Nitrates are not subject to biodegradation mechanisms and, therefore, serve as a good indicator of dilution. Refer to the other sections of this CAP for the discussion regarding dilution and nitrate trends. In contrast, the available data do not support biodegradation as a mechanism that is measurable or observable. These issues are presented in this section for purposes of comparison or explanation so that biodegradation is put into proper perspective as a negligible contributor to natural attenuation at this site.

It is also important to point out that biodegradation of the COCs is only applicable for purposes of this CAP under the natural conditions found at this site. Various researchers have documented the biodegradation of some of these compounds under very specific, artificial laboratory conditions, but that information has little relevance to the topic of biodegradation as a component of natural attenuation at this site, except to define the conditions that may be necessary for biodegradation to occur. Only the potential for biodegradation under the conditions found at this site are applicable for purposes of the CAP. The types of conditions necessary to observe biodegradation in a laboratory setting typically may include, but are not limited to the following:



- aerobic or anaerobic conditions (one condition or the other may be necessary to facilitate biodegradation for some compounds)
- specific microorganism addition or supplementation
- presence or absence of other compounds, including supplemental carbon sources
- absence of inhibiting chemistry (i.e. nitrates can inhibit RDX biodegradation, Freedman, 1998)

In addition to the conditions necessary for biodegradation to occur, other conditions that may enhance degradation in a laboratory environment include, but are not limited to the following:

- mechanically mixed or homogenized laboratory environment (as opposed to natural, in-situ environment)
- higher than natural temperatures

7.2.4.6 *Individual COCs*

7.2.4.6.1 General

In considering the potential for biological breakdown of the parent COCs, specific criteria were applied. A thorough review of the literature on this topic, for the compounds of interest, was performed to identify potential or theorized breakdown products. The subject of explosives breakdown and analysis for known or hypothesized breakdown compounds is a developing and complex field of research. A majority of this research has been sponsored by the United States, related to its military facilities. This discussion is based on a review of that information and, where appropriate, offers a condensation of the available information and draws conclusions consistent with the state of knowledge base and analytical techniques.

Two related concepts were applied in the identification of potential breakdown products for parent COCs for this site. First, the literature must confirm the generation of the breakdown product under the conditions consistent with the natural environment of the site. Some breakdown products identified in the literature are only theorized or have been generated under very specific artificial conditions. Breakdown products identified in only either of these two categories are not included as possible breakdown products for the site. The literature must also offer confirmation of breakdown products under conditions reasonably similar to the site conditions to be considered as a potential breakdown product for this site. Confirmation must include an experimental design that isolates the conditions of interest using analytical methods that are reasonably verifiable. Secondly, there must be established laboratory analytical methods and commercially available chemical reference standards for the analysis of the breakdown products for the breakdown products to be considered in this CAP. In other words, there must be readily available means to analyze for these compounds in order to be investigated as potential breakdown products in this CAP. Some of the literature information on breakdown products use novel analytical methods that fall into the category of research work and are not readily translatable to the realm of commercial laboratory work. In the instance of



RDX, as discussed below, research work was conducted by our project team to examine the potential presence of certain theorized breakdown products at this site. This work was specifically requested by the DEQ.

These criteria were applied in assessing potential biodegradation products for the parent COCs.

7.2.4.6.2 RDX and HMX

The literature is not straightforward with respect to identifying breakdown products for RDX and HMX. A brief summary of the history on this topic is presented as follows. In 1981 McCormick, et. al., published a paper on the breakdown of RDX. In that paper, numerous intermediate and endpoint products were postulated. For many years, this paper, and the postulated breakdown pathway, was the reference point for the majority of other research on this topic.

Recently, a reference entitled, "Biodegradation of Nitroaromatic Compounds and Explosives", edited by Jim Spain, et. al. (2000), contains a chapter in which a thorough treatment of the subject of RDX and HMX degradation is presented. A good discussion of the relevance of the McCormick work is discussed. In this current review of the state of the research and from a general review of the literature, it is apparent that the influence of the early McCormick work is pervasive in the interpretation of other studies, resulting in the perpetuation of certain assumptions that have not been confirmed. Refer to the chapter in Spain's book, authored by Hawari, for a discussion of the most current thinking regarding the degradation of RDX and HMX. This chapter contains references to over 100 technical papers. A number of those papers are also included in the reference list for this section of the CAP.

In summary, much of what has been previously postulated as potential breakdown products and pathways for RDX is now questioned. At a minimum, there is disagreement in the scientific community with regard to the breakdown products, at least beyond the first order products (MNRDX, DNRDX and TNRDX). For instance, direct ring cleavage of RDX has been reported. There seems to be agreement that, under anaerobic conditions, the generation of MNRDX is possible. Beyond MNRDX, there are alternative pathways described that are probably very much dependant on environmental conditions. In some cases, direct ring cleavage of MNRDX is postulated or reported. In other cases, the sequential formation of DNRDX and TNRDX is reported. Beyond formation of these so-called first order degradation products or nitroso compounds, the literature is in disagreement and the research is on-going. Therefore, for this CAP, potential breakdown products of RDX will be limited to the three RDX nitroso compounds (MNRDX, DNRDX & TNRDX).

Standard laboratory methods do not exist for the first order RDX nitroso compounds; however, it is our understanding that at least one commercial laboratory is providing limited analytical services for the detection of the first order RDX nitroso compounds



using non-standard methods. Analytical standards for these compounds are not known to be commercially available, but can be obtained from one or two research and development sources. Therefore, analysis for these compounds is possible, but it is by non-standard methods.

At the request of DEQ, a program was established to investigate the potential for the presence of RDX breakdown products in the regional aquifer. Three wells (MW-11D, UP&L and Young) were selected as representing the highest RDX concentrations observed in area ground water. Potential breakdown products were selected (formaldehyde, hydrazine and the nitroso products – MNRDX, DNRDX & TNRDX) and tests were run to evaluate their presence. There was a verbal agreement that if these compounds were not found to be present at levels above their detection limits, the issue of assessing the potential for presence of breakdown products would be concluded. This work has been documented in various previous reports (ERM, 2000a and 2000b) and only a brief summary is presented herein. None of the potential breakdown products were reported at levels above their analytical detection limits. It is notable that formaldehyde and hydrazine may not be endpoint products in the biodegradation of RDX under environmental conditions. In the case of hydrazine and related compounds, an extensive effort to develop suitable methods for analysis at very low levels was unsuccessful. However, in addition to the uncertainty that these compounds are actually products in the breakdown pathway for RDX, it has also been confirmed in the literature that the hydrazine compounds are extremely unstable at low concentrations in the environment and are not expected to persist in the environment, even if generated (Spain, 2000; McCormick, 1984; Human Health Perspective on Environmental Exposure to Hydrazines, 1998; Hydrazine Health and Safety Guide, 1991). The end result of these investigations was to conclude that there is no evidence of RDX biodegradation in the regional aquifer at this site at levels that are detectable, at about a one part per billion detection limit. Therefore, no further evaluation of the biodegradation of RDX or RDX biodegradation products is proposed in this CAP.

The compound HMX is structurally similar to RDX, but less work has been reported in the literature with respect to the generation of HMX breakdown products. A March 1998 study conducted by the Amarillo National Resource Center for Plutonium specifically identifies four nitroso compounds (mono, di, tri and tetra), as the first order breakdown products of HMX (Card, 1998). In further review of this topic, it has been identified that there are two isomers of the di-nitroso HMX. Therefore, there are five distinct possible compounds. However, the literature review did not identify a specific, published laboratory procedure or source of analytical standards for the analysis of HMX nitroso compounds. It is our understanding that there is no commercial source for these standards. Given the general similarities between RDX and HMX, the much lower concentrations of HMX compared to RDX, and the lack of standard laboratory methods or standards for the HMX nitroso compounds, no further assessment of the presence of these compounds is proposed.



7.2.4.6.3 TNT and DNTs

The literature has a relatively large amount of information on degradation of TNT and DNT compounds in comparison to the other CEMs. This is probably due to the common occurrence of these compounds on military sites. The information regarding breakdown products of TNT and DNT is not presented in this CAP, however, due to the fact that these compounds have not been confirmed as present in the regional aquifer. As described in Section 6.8 of this CAP the CEMs 2,4,6-TNT and 2,6-DNT have only been detected sporadically at several locations in the regional aquifer. These two CEMs and 2,4-DNT were not detected in the regional aquifer during sampling performed in 2001. Therefore, there is no practical relevance to this CAP regarding the potential for the generation of breakdown products of these compounds. The concentrations of potential breakdown products would be so low as to make them impossible to identify at the available detection limits. Therefore, potential breakdown products of TNT and DNT are not considered further in this CAP and are not considered to be of any practical relevance.

7.2.4.6.4 Nitrate Esters

The nitrate esters include PETN, EGDN, DEGDN, TEGDN, TMETN and BTTN. Although included within this category, NG is not included in this discussion since it is not detected in the regional aquifer. Due to the structural similarities of these compounds they tend to be treated as a group in the literature. There is relatively little information available for this group of compounds and the literature that is available tends to be older than for some of the other CEMs. It would seem that this may very well be due to: a lower occurrence of these compounds on sites; the relative simplicity of degradation of the nitrate ester compounds in comparison to other CEMs; and, the reported low toxicity of the degradation products (Cornell, 1981; Govind, 1994; Kaplan, 1981; Shepodd, 1997; Spanggord, 1985; Spanggord, 1987).

Published biodegradation studies for nitrate esters are for laboratory experiments. No in-situ studies were identified. As such, the studies tend to establish artificial conditions that support degradation of these compounds. It is important to keep in mind that in the natural environment, biodegradation may not occur as quickly or at all in some instances, due to the lack of some key facilitating factor such as suitable bacteria. Nevertheless, the biodegradation of nitrate esters follows a path of nitrate reduction and resulting in ultimate mineralization. Degradation is reported under both aerobic and anaerobic conditions.

It is generally reported that the degradation products of nitrate esters have a lower toxicity than the parent compounds. Considering that the nitrate esters do not currently exceed the CACLs proposed for this CAP, the potential degradation products are of no practical relevance with respect to their toxicity for this CAP.



Overall, the potential for biodegradation of nitrate esters is shown to have no practical relevance for purposes of this CAP. What is relevant is simply the disappearance of the parent compounds. Dilution is still thought to be the dominant mechanism for reductions in the concentrations of the nitrate esters. To the extent that there is any biodegradation that occurs, the intermediate or endpoint products formed are reported to be less toxic than the parent compounds and not of concern. Therefore, no further discussion on the biodegradation of the nitrate esters is presented and no further evaluation of nitrate ester biodegradation products is proposed in this CAP.

7.2.5 Facilitated Transport

According to McGrath (1995), facilitated transport (FT) is a general term for a group of mechanisms by which the mobility of chemical compounds is increased. McGrath proposes several mechanisms of FT including cosolvation, macromolecules, colloids, micelles and emulsions. With the exception of cosolvation, the degree to which these mechanisms may enhance the subsurface transport of CEMs is not well understood and will not be discussed further.

Cosolvation is the tendency for the solubility of organic compounds to increase with dissolved organic carbon concentration. For example, PETN has a very low solubility in water whereas it is readily soluble in polar organic solvents such as acetone (Etnier, 1986). Cosolvation could be a viable fate mechanism in the immediate vicinity of potential initial infiltration areas where acetone was used in the processing of PETN, RDX and HMX or where ethyl acetate was used to dissolve TNT. It should be noted that as the solvent is diluted with water, either in the discharge stream or in ground water, the CEM solubility would decrease, approaching the solubility of that particular CEM in water. Since organic solvents have not been detected in regional aquifer, it is unlikely that cosolvation is a significant factor on the large scale. Cosolvation is not practically relevant to this CAP.

7.2.6 Volatilization

Volatilization is the phase change of a compound from a solid or a liquid into a gas. The tendency for a compound to volatilize is determined by a physical constant called the Henry's Law constant. A higher volatilization potential corresponds to a higher Henry's Law constant. According to McGrath (1995), organic compounds that have a Henry's Law Constant (K_H) of less than 10^{-7} atm-m³/mole are considered to be essentially non-volatile. Those that have a K_H of between 10^{-7} and 10^{-3} atm-m³/mole are considered semi-volatile and compounds that have a K_H higher than 10^{-3} atm-m³/mole are considered volatile organics. Nitrate and CEMs have low Henry's Law constants and do not readily volatilize. Volatilization is not considered to be a significant environmental fate and transport factor for solutes present in ground water within the study area and is not practically relevant to this CAP.



7.2.7 Photolysis

Photolysis is the degradation of constituents in the presence of light and, in the case of the natural environment, sunlight is the light source of interest. This CAP is interested in the fate of constituents in a ground water environment, where sunlight is absent. Therefore, photolysis is not considered to be a potential mechanism of natural attenuation in the ground water environment and is not discussed in detail in this CAP.

In the RFI investigation, constituents have been detected in soils at the very surface, which are exposed to sunlight and, therefore, might undergo photolysis. In order for soils to be exposed to sunlight, they would need to be located within the upper inch of soils. Current evidence suggests that soils remaining on the site do not contribute as a continuing source in any observable way to concentrations of constituents in the regional aquifer. Therefore, any potential photolysis of very shallow soils is not considered to be a mechanism of concern in the evaluation of ground water quality in the regional aquifer.

Historical discharges to the ground were subjected to sunlight and photolysis may have been a degradation mechanism at that time. However, those practices were discontinued many years ago and are not of current interest. If photolytic breakdown products were generated historically and they have not further degraded, then they should be observed in the current monitoring program. The efforts to investigate breakdown products for individual constituents discussed below are applicable to the historical potential for photolytic breakdown of these compounds as well as the potential for biodegradation. For RDX and HMX, the only breakdown product that is confirmed as a uniquely photolytic breakdown product is formaldehyde and specific analyses for formaldehyde did not find this compound present at detectable levels, in this case at concentrations at or above 1 µg/L.

Based on the discussion above, photolysis is not considered to be a mechanism that has any relevant contribution to ground water quality in the regional aquifer and is not discussed further in this CAP.

7.2.8 Hydrolysis

Hydrolysis is a chemical reaction between a chemical compound and water that results in the disassociation of that chemical compound. Hydrolysis is an abiotic process. There is no evidence that hydrolysis plays a significant role in the transformation of CEMs under environmental conditions (Burrows et. al., 1989). McGrath (1995) indicates that most CEMs and related compounds are resistant to hydrolysis except in high pH (pH >9) environments. Hydrolysis is not considered to be a viable environmental fate and transport mechanism at in the study area and will not be discussed further in this CAP.



7.2.9 Other Abiotic Processes

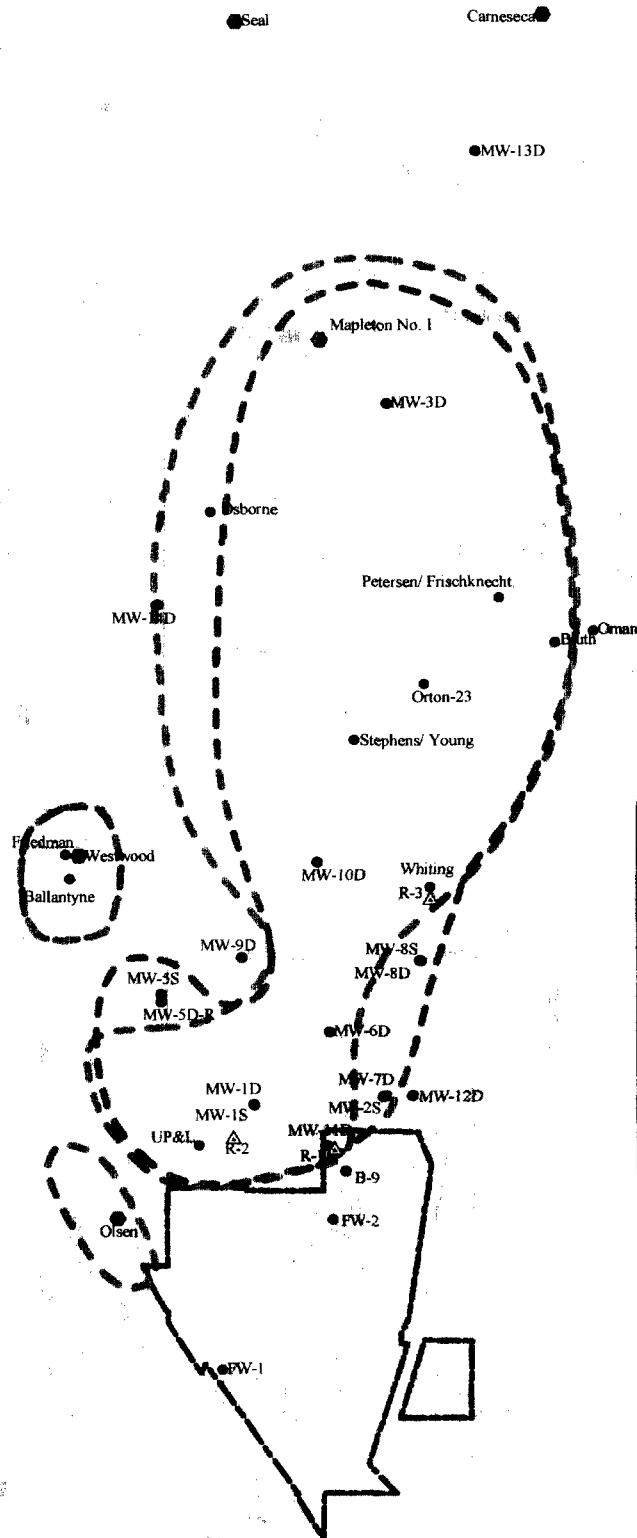
McGrath (1995) reports that several other abiotic processes may affect the environmental fate and transport of CEMs. These include acid-base dissociation reactions, oxidation-reduction (redox) reactions and irreversible surface reactions. According to McGrath, little data is available for CEMs with respect to these abiotic processes. McGrath notes that experimental evidence suggests that TNT may undergo abiotic reduction in soils having relatively abundant organics or anaerobic microenvironments, whereas RDX and HMX appear to be much less susceptible to abiotic reduction. TNT and other nitroaromatic CEMs may also undergo irreversible sorption to soils through the formation of covalent bonds between the chemical compound and the humic structure of the soil. The lack of research and data makes it impossible to know what affect, if any, these additional abiotic processes may have on the environmental fate and behavior of CEMs in the study area.

7.3 General Area of Solute Distribution

Figure 7-1 is a map of the study area that illustrates the approximate or inferred extent of solute migration within the regional aquifer based on water quality data collected in the year 2001. The area of nitrate-nitrogen detections at or above 5 mg/L is outlined in light green and the area of RDX detections above the RDX method detection limit (MDL) is in blue. Water quality conditions within the regional aquifer vary, therefore certain wells located within the approximate area of impact may not have nitrate-nitrogen concentrations that are greater than 5 mg/L or RDX may not be present above the MDL. For the purpose of this simple presentation, no comparison to proposed corrective action concentration limits (CACs) for nitrate-nitrogen or RDX are made or implied. Ground water impacts have only been identified in the deep regional aquifer in the northern and western regions of the study area. Therefore, the area of impact is inferred based on wells that are open to the deep regional aquifer in these locations (MW-3D, MW-13D, MW-14D, Seal, Mapleton No. 1, Westwood). Impacts have not been observed in the shallow regional aquifer as evidenced by samples collected from MW-5S, Osborne, Ballantyne, Friedman, and Cobia, even though these wells are located within the approximate solute distribution boundaries. Water quality data are summarized in Appendix B and are presented visually in Figures 7-2 through 7-11 found in the map pockets at the end of this document.

This figure shows that the affected region, as described by the approximate boundaries in the previous paragraph, is elongated in a northerly direction along the mountain front, extending a short distance north of the Mapleton No. 1 well. The long axis (northerly orientation) of the affected zone is approximately 3.5 miles for CEMs and 3.75 miles for nitrate. The short axis (westerly orientation) of the affected zone is approximately 1 mile for RDX and 1.5 miles for nitrate. The affected area has length to width ratio of approximately 3:1. The distribution of solutes suggests that most of the ground water flow is directed to the north-northeast. The distribution of both nitrate and RDX also indicates some solute migration to the west in the vicinity of MW-1S, MW-1D, UP&L and MW-5D. Although accurate predictions about ground water flow paths cannot be



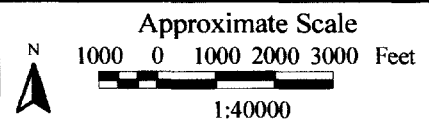


EXPLANATION

- Approximate NO₃-N > 5 mg/L
- Approximate RDX > MDL
- △ Extraction Wells
- Monitoring Wells
- Private Wells
- Municipal Wells
- EBCo Property Boundary

NOTES:

- * Only wells where water quality data are available for 2001 are shown.
- * Nitrate-nitrogen has not been detected above 5 mg/L in Friedman, Ballantyne, and Osborne.
- * RDX has not been detected in R-2 and MW-5S.



CHARTER OAK
4505 South Wasatch Blvd., Suite 360
Salt Lake City, Utah 84124
TEL: (801) 277-6150
FAX: (801) 277-6151

Nitrate-Nitrogen and RDX Approximate Distribution Map 2001

FIGURE 7-1

made from the available water level data, the solute distribution is consistent with the approximate water level elevation contours presented in Figure 6-19. The foregoing is consistent with the former wastewater management area located in the northeast portion of the Plant site where the wastewater dispersion area, north impoundment and former storage ponds were located. The nitrate distribution also suggests a second source area in the northwest portion of the Plant site where acids from nitroglycerin production and acids and wastewater from nitrostarch operations were managed in the area of the northwest impoundment. Nitrate impacts at the Olsen well may be attributable to this wastewater management area.

7.4 Site Specific Chemical Data

Details regarding the constituents of concern present in the study area are provided herein. Practically relevant environmental fate and transport mechanisms are also discussed for each compound. Biodegradation, photolysis, cosolvation, hydrolysis and other abiotic processes that are not practically relevant to the CAP are not discussed. Information about the toxicity of these compounds is provided in Appendix E and is not discussed further in the body of this report.

Distribution and trend maps are presented that illustrate the concentration trends and distribution of nitrate-nitrogen and selected CEMs present in ground water. Figures 7-2 through 7-11 are located in the map pockets at the end of this document. The color scheme of the individual bar charts indicates the relative location and depth of the wells. Dark purple indicates wells located on the topographic bench in the southeast side of the study area and that are generally within the foothills recharge area. Magenta represents the wells that are open to the shallow regional aquifer underlying the Mapleton Bench area or wells that have screen intervals at similar elevations. Light purple represents wells that are open to the deep regional aquifer that underlies the Mapleton Bench or wells that have screen intervals at similar elevations. Orange indicates wells that are open to the regional bedrock aquifer. Make note of the vertical scale of the charts as they vary for several of the compounds. The vertical scale for the trend charts on the Nitrate-nitrogen map is from 0 to 60 mg/L. The vertical scale for RDX and total specialty nitrate esters is from 0 to 60 µg/L and the vertical scale on the remainder of the maps is 0 to 16 µg/L. Tables containing the raw data used to prepare the Distribution and Trend maps are provided in Appendix B.

Trend analyses in this report were performed on nitrate-nitrogen and CEM concentrations in water samples collected from monitoring wells and recovery wells. The Mann-Kendall test method was used in the trend analyses. The Mann-Kendall test method is a non-parametric test and is performed using a program listed in Appendix B of Gilbert (1987). As mentioned in Gilbert (p.202, 1987), at least ten data points are needed for application of the Mann-Kendall test.

In Mann-Kendall analyses, the hypothesis of no trend against the alternative hypothesis of either a downward trend or an upward trend (two-sided test) is tested at a significance



level of $\alpha = 0.05$. In order to reject the hypothesis of no trend at $\alpha = 0.05$ for a two-sided test, the absolute value of the Z statistic calculated by the Mann-Kendall method must be greater than or equal to 1.960. By using a significance level of $\alpha = 0.05$, it is anticipated that there will be a five percent chance of rejecting the hypothesis of no trend when it is true.

The ensuing discussions regarding concentrations trends reflect the results of the Mann-Kendall trend analyses. Mann-Kendall trend analyses results for these data are provided in Appendix C.

7.4.1 Nitrate-Nitrogen

The potential source of nitrates in ground water from Plant sources were the discharges associated with the formulation of nitroglycerin, nitrostarch, specialty nitrates and PETN. The volume of releases to the environment between 1941 and 1991 are not known. Dilute nitric acid (approximately 20%) was stored in lined surface impoundments from late-1980 until June 1986 when this storage practice ceased following a second breach in the liner that resulted in the release of dilute nitric acid. Other than some infrequent, small (tens of gallons) releases during acid handling activities and which were cleaned up immediately, there have been no known discharges of acid to the ground surface since 1991. Potential off-site sources of nitrates were identified and discussed in section 6.7.4 of this report.

7.4.1.1 Environmental Fate and Transport

Nitrate is very soluble and mobile in ground water. Nitrate is a conservative solute that does not adsorb to the aquifer matrix and does not degrade appreciably in the subsurface environment. Nitrates will generally migrate as a result of advective-dispersive processes. Nitrate probably has diffused into fine-grained deposits in the aquifer system.

7.4.1.2 Distribution and Concentration Trends

Figure 7-2 illustrates the distribution and concentration trends for nitrate-nitrogen from selected wells from 1989 until the fourth quarter of 2001. Several observations can be made upon review of the nitrate-nitrogen distribution and trend map.

- Nitrates are widely distributed reflecting its high mobility and long source history.
- Based on the available data, the highest nitrate-nitrogen concentrations were historically identified within and adjacent to the Crowd Canyon alluvium. This is close to the known source area present in the northeast corner of the Plant. During the 1970's high concentrations of nitrate were also identified in the Olsen well. Operational practices in the northwest corner of the Plant site probably



contributed to the elevated nitrate-nitrogen concentrations detected in the Olsen well.

- Nitrate-nitrogen concentrations have downward trends in wells located close to the Plant and within or proximal to the Crowd Canyon alluvium. These declining trends reflect the cessation of discharges in 1991, as well as dilution due to higher amounts of recharge to this area between 1994 and 1998. The nitrate-nitrogen concentrations in many of the wells in this area appear to be leveling off (asymptotic).
- In accordance with the Mann-Kendall test, Nitrate-nitrogen concentrations at MW-10D and MW-14D exhibit upward trends, which are being addressed by the installation of additional monitoring wells as described in Section 12.6.2 of this CAP. Wells further to the north (i.e. Frischknecht, MW-3D, and Mapleton No. 1) generally have either no trends or downward trends.
- Nitrate-nitrogen concentrations at the Evans/Young well have a downward trend but are higher than nitrate-nitrogen concentrations in other wells screened at similar depths in the area. The reason for this phenomenon remains undetermined but might indicate an alternative nitrate source or an unidentified subsurface feature affecting the local ground water flow and solute movement. The Booth well, located due south of the Evans and Young wells is open to a permeable sand and gravel unit that may be a buried stream channel associated with the Crowd Canyon Alluvium. In 1989, nitrate-nitrogen concentrations in the Booth well were similar to that of the Evans and Young wells. The Booth well has not been sampled since 1989.
- An upward trend in nitrate-nitrogen concentrations is also observed at FW-2. However, nitrate-nitrogen concentrations in FW-2 were below 1.5 mg/L at the end of year 2001.
- The appearance of low concentrations of nitrate at MW-8S in 1998 is coincident with the rise in water levels in this area from 1994 through 1998. This may reflect a mobilization of nitrates during the saturation of previously unsaturated units or a change in ground water flow paths caused by the increase in water levels. Nitrate-nitrogen concentrations in MW-8S have declined since 1999.
- The Ballantyne, Friedman, Osborne, Clark and Cobia wells and MW-5S have relatively low nitrate-nitrogen concentrations.
- Nitrate-nitrogen concentrations at Seal, Carneseca, Hjorth, MW-9D and MW-13D have remained consistently low or below detection limits. Furthermore, sample data for nitrate-nitrogen concentrations at the Mapleton No.1 and Westwood wells have never exceeded the MCL of 10 mg/L.
- Nitrate-nitrogen data from B-9, FW-1 and FW-2 are relatively low.



- Nitrate-nitrogen data from MW-12D and Oman indicate that the bedrock aquifer provides an eastern boundary for water quality impacts.
- With the exception of Evans/Young, MW-10D and R-2, nitrate-nitrogen concentrations in wells throughout the study area were below the 10 mg/L MCL during the third and fourth quarters of 2001.

7.4.2 RDX

RDX was processed at the site from approximately 1971 until 1986. From 1971 to the mid to late 1970's, RDX operations, involving the processing of Comp A3 took place only in the northeast manufacturing complex (SWMU 30). Comp B processing began in 1980 at the present location of the PETN nitrator (SWMU 19). Throughout the entire period of RDX processing, discharges from waxing operations, crystallization, re-crystallization and acetone recovery processes were directed to the wastewater conveyance channel and wastewater dispersion area.

7.4.2.1 *Environmental Fate and Transport*

The aqueous solubility of RDX is between approximately 29 and 42 mg/L for the ground water temperatures present within the regional aquifer (see Table 7-1). The Log K_{oc} estimates for RDX indicate that RDX has a moderate tendency to sorb to aquifer materials relative to the other CEMs. Sorption processes may be an important consideration for the long-term fate of RDX in the regional aquifer. RDX migration in ground water will be retarded to some degree due to sorption processes. Some diffusion of RDX into finer grained deposits is expected.

7.4.2.2 *Distribution and Concentration Trends*

RDX data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-3. Several observations can be made upon review of the RDX distribution and trend map.

- The distribution pattern reflects the relative mobility of RDX and the production history from 1971 through 1986.
- RDX is not found at Olsen, Westwood and MW-14D where nitrate-nitrate concentrations exceeded 5 mg/L during 2001. RDX is also not present in MW-5S, MW-9D, MW-12D, MW-13D, B-9, FW-1, FW-2, Seal and Carneseca.



- As observed with the nitrate-nitrogen data, the highest RDX concentrations were historically detected within and adjacent to the Crowd Canyon alluvium. This is close to the source area formerly present in the northeast corner of the Plant.
- Downward trends in RDX concentrations are present in wells open to the regional aquifer in the area of the Crowd Canyon alluvium and are also present at MW-8D and Whiting. With the exception of MW-8D, declines in RDX concentration are not as steep as observed for nitrate. This probably reflects a different source history. Also, RDX will adsorb to aquifer materials, therefore retarding its movement relative to that of nitrate. Desorption of RDX from aquifer materials into clean recharging ground water may also account for the slower observed declines in concentration.
- No trends in RDX concentrations are present at the locations of MW-1S, MW-1D and UP&L.
- Upward RDX concentration trends are evident at MW-10D, Evans/Young and MW-3D. No trends are present at Frischknecht and Mapleton No. 1. RDX concentrations in Mapleton No. 1 have remained relatively low ($<4 \mu\text{g/L}$).
- RDX concentrations in MW-5D show an upward trend, although concentrations remain relatively low ($<4 \mu\text{g/L}$).
- The appearance and disappearance of RDX at MW-8S during 1998 may reflect desorption or diffusion of RDX from aquifer materials due to the saturation of previously unsaturated deposits. It could also represent changing ground water flow paths resulting from the increase in water levels in the Crowd Canyon alluvium.
- During 2001, RDX concentrations exceed the interim ground water quality goal of $2 \mu\text{g/L}$ at nineteen of the thirty three wells where CEMs were analyzed as part of the ground water monitoring program.

7.4.3 HMX

Due to their similar chemical structures and manufacturing process, HMX is a common impurity of RDX. RDX containing up to 9% by weight of HMX is considered typical and acceptable for production. HMX was also processed at the site for a short period of time during the mid-1980's. Processing steps included acetone re-crystallization and packing. HMX operations were conducted in SWMU 30 and operations resulted in discharges to the wastewater conveyance channel and dispersion area (SWMU 1).



7.4.3.1 Environmental Fate and Transport

The aqueous solubility of HMX is between approximately 1.2 and 2.6 mg/L for the ground water temperatures present within the regional aquifer (see Table 7-1). The Log K_{oc} estimates for HMX indicate that HMX has a relatively strong tendency to sorb to aquifer materials when compared to the other CEMs. Sorption processes are probably an important consideration for the long-term fate of HMX in the regional aquifer. HMX migration in ground water will be retarded due to sorption processes relative to many other CEMs. Some diffusion of HMX into finer grained deposits is expected.

7.4.3.2 Distribution and Concentration Trends

HMX data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-4. Several observations can be made upon review of the HMX distribution and trend map.

- The overall distribution of HMX is more limited than that of nitrate, RDX and several of the specialty nitrate compounds. This may reflect a combination of factors including its relatively small production volume, lower solubility and relatively greater propensity to adsorb to aquifer materials.
- The highest concentrations of HMX are found in wells within or adjacent to the Crowd Canyon alluvium. This is close to the likely source area in the northeast corner of the Plant.
- Downward trends in HMX concentrations are present at MW-1S, MW-2S, MW-6D, MW-7D, MW-8D, MW-11D and R-1.
- The Whiting, Evans/Young and Orton-23 wells have the only upward trends in HMX concentrations.
- HMX has not been detected at Bluth, MW-3D, MW-5D, MW-10D and Mapleton No. 1 and only very low concentrations ($<4 \mu\text{g/L}$) have been detected at some of the more distal wells (Frischknecht, Evans/Young).
- The HMX distribution is consistent with the northerly ground water migration pathway with some westward migration in the vicinity MW-1S and UP&L. The lack of detections at MW-3D, MW-5D and Mapleton No. 1 probably reflect retardation along the travel path due to sorption.
- The appearance and disappearance of HMX at MW-8S during 1998 may reflect desorption or diffusion of HMX from aquifer materials due to the saturation of previously unsaturated materials. It could also represent changing ground water



flow paths resulting from the increase in water levels in the Crowd Canyon alluvium.

- Based upon the available water quality data, HMX concentrations have been and remain well below the proposed CACL of 400 µg/L.

7.4.4 2,4,6-TNT

TNT was not manufactured at the site. TNT is purchased and added to PETN in melting pots to produce Pentolite boosters, a process which generates minimal discharges. These discharges would have been at locations where soils have not been found to be impacted in the RFI investigation. Since 1991, these discharges have been managed in the wastewater treatment system and discharged to the sanitary sewer. TNT is also a component of Comp B, which is a blend of RDX and TNT. During Comp B processing, the Comp B was submerged in ethyl acetate, which dissolves TNT but not RDX. The TNT was crystallized and either used in Pentolite boosters or destroyed by open burning. There were no known systematic discharges associated with TNT operations.

7.4.4.1 *Environmental Fate and Transport*

The aqueous solubility of 2,4,6-TNT is between approximately 110 and 150 mg/L for the ground water temperatures present within the regional aquifer (see Table 7-1). The Log K_{oc} estimates for 2,4,6-TNT indicate that 2,4,6-TNT has a high tendency to sorb to aquifer materials relative to the other CEMs. McGrath (1995) indicates that some soil studies have shown that 2,4,6-TNT may irreversibly bind to the soil matrix therefore permanently removing 2,4,6-TNT from ground water. Sorption processes may be an important consideration for the long-term fate of 2,4,6-TNT in the regional aquifer. 2,4,6-TNT migration in ground water will be retarded significantly due to sorption processes relative to many other CEMs.

7.4.4.2 *Distribution and Concentration Trends*

Over the 1995 to 2001 monitoring period, 2,4,6-TNT has only been reported as detected in the regional aquifer a total of four times and from two different wells (MW-10D, 0.35 and 0.39 µg/L; FW-1, 2.16 and 1.82 µg/L) and was not detected at all in 2001. Considering its limited detections, it is unknown if these reported detections are real or artifacts of the analytical method. The limited apparent detections in the regional aquifer indicate the 2,4,6-TNT is not widely distributed at concentrations above the MDL. In light of the foregoing, no distribution and trend map is provided for 2,4,6-TNT.

The lack of significant, if any, 2,4,6-TNT in the regional aquifer most likely reflects the limited use of this compound in production activities.



7.4.5 2,4-DNT and 2,6-DNT

The CEMs 2,4-DNT and 2,6-DNT were not produced at the Plant. Both 2,4-DNT and 2,6-DNT are found as impurities in purified (military grade) TNT (Borrows, et. al., 1989). A 1958 map illustrating the layout of the Plant during the period of nitroglycerin production indicates the presence of a DNT heater house. This suggests that DNT may have been used during nitroglycerin production; however, the nature of its potential use remains undetermined.

7.4.5.1 *Environmental Fate and Transport*

The aqueous solubilities of 2,4-DNT and 2,6-DNT are less than 270 mg/L and 206 mg/L, respectively for the ground water temperatures present within the regional aquifer (see Table 7-1). The Log K_{oc} estimates for 2,4-DNT and 2,6-DNT indicate that these compounds have a moderate tendency to sorb to aquifer materials relative to the other CEMs. Given the very limited detections of 2,6-DNT and lack of detections of 2,4-DNT in ground water, sorption processes are probably not an important consideration for the long-term fate of these compounds in the regional aquifer. Migration of 2,4-DNT and 2,6-DNT in ground water will not be retarded significantly due to sorption processes relative to many other CEMs.

7.4.5.2 *Distribution and Concentration Trends*

Over the 1995 to 2001 monitoring period, the CEM 2,6-DNT has only been reported as detected in the regional aquifer a total of four times and from four different wells (UP&L, 0.44 µg/L; MW-1S 0.26 µg/L; R-1, 0.47 µg/L; R-3, 0.23 µg/L) and was not detected at all in 2001. The CEM 2,4-DNT has not been detected in ground water since monitoring for this compound began in the first quarter of 2001. Given the limited and variable detections of 2,6-DNT, it is unknown if these reported detections are real or artifacts of the analytical method. The limited apparent detections in the regional aquifer indicate that 2,6-DNT is not widely distributed at concentrations above the MDL. In light of the foregoing, no distribution and trend maps are provided for 2,4-DNT and 2,6-DNT.

The lack of 2,4-DNT and of significant, if any, 2,6-DNT in the regional aquifer most likely reflects the fact that 2,4-DNT and 2,6-DNT are probably only present as an impurity of TNT, and TNT use was limited. Based on the RFI data, only low concentrations of 2,4-DNT and 2,6-DNT have been detected in on-site soils at a few locations.

7.4.6 EGDN

The NG produced at the Plant from 1941 to 1963 was a mixture of NG and EGDN. Wastewater discharges from NG/EGDN production during this time period would have



contained both NG and EGDN. Wastewater discharges from NG/EGDN production were directed to the wastewater conveyance channel and wastewater dispersion area. Based on the aqueous solubility of these compounds, the NG to EGDN ratio in the wastewater discharge would have been approximately one to four (1:4).

EGDN was also manufactured during later production activities at the Plant as a specialty nitrate compound. Specialty nitrates production began in 1976 and was terminated in 1991. All the specialty nitrates were formulated in the old nitroglycerin nitration building. Wastewater discharges from specialty nitrates production were directed to the wastewater conveyance channel, North Impoundment (after its construction in 1987) and the wastewater dispersion area (prior to 1987). Waste acids from specialty nitrates production were burned in the burn pit prior to about 1987. After 1987, waste acids were collected in tanks, concentrated and removed from the site and recycled by a third party. The exact history of the production of specific specialty nitrate compounds remains undetermined. Each compound was produced in batches based on customer demand. The original process was developed for TMETN production in 1976. Plant records indicate that there were several large contracts for EGDN during the mid- to late 1980's.

7.4.6.1 Environmental Fate and Transport

The aqueous solubility of EGDN is reported to be 5,600 mg/L at 25°C, therefore the aqueous solubility of this compound is less than 5,600 mg/L at the ground water temperatures present in the regional aquifer (see Table 7-1). EGDN has the highest aqueous solubility of all of the CEMs present in the regional aquifer. The estimated Log K_{oc} values for EGDN are some of the lowest for the CEMs indicating that sorption is not an important fate and transport factor for this compound. This is consistent with the observation that EGDN is the first CEM to break through the GAC treatment systems. Some diffusion of EGDN into fine grained deposits is anticipated.

7.4.6.2 Distribution and Concentration Trends

EGDN data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-5. Several observations can be made upon review of the EGDN distribution and trend map.

- EGDN is found in most wells where RDX is present. The distribution pattern probably reflects high solubility and mobility of EGDN.
- The highest EGDN concentrations were historically detected within and adjacent to the Crowd Canyon alluvium. This is close to the source area formerly located in the northeast corner of the Plant.



- Downward trends in EGDN concentrations are present in MW-2S, MW-7D, MW-11D, Mapleton No. 1 and R-1.
- No upward trends in EGDN concentration are present at the other wells where EGDN has been detected.
- EGDN did not appear in MW-5D until June 1998.
- EGDN is the only CEM present in recovery well R-2. R-2 screens very fine sand and silt deposits in the shallow regional aquifer. Nitrates in R-2 exceed 10 mg/L. Therefore, the two most soluble constituents and two of the constituents that are least likely to adsorb to aquifer materials are present in these fine-grained deposits.
- Sporadic low detections of EGDN along the eastern boundary of the study area are different than the distribution observed for RDX and other specialty nitrate compounds (DEGDN and TMETN).
- EGDN concentrations have never exceeded the proposed CACL of 52 µg/L at any of the monitoring wells and are not expected to do so in the future.

7.4.7 DEGDN

DEGDN production is the same as described for EGDN as a specialty nitrate. Although the exact dates of DEGDN production remain undetermined, specialty nitrates were produced at the Plant from 1976 to 1991. Plant records indicate that there were several large contracts for DEGDN during the mid- to late 1980's.

7.4.7.1 Environmental Fate and Transport

The aqueous solubility of DEGDN is reported to be 4,000 mg/L at 20°C, therefore the aqueous solubility of this compound is less than 4,000 mg/L at the ground water temperatures present in the regional aquifer (see Table 7-1). The Log K_{oc} estimates for DEGDN are similar to those for RDX indicating that sorption should be considered to be an important fate and transport factor for this compound. Some diffusion of DEGDN into fine grained deposits is anticipated.

7.4.7.2 Distribution and Concentration Trends

DEGDN data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-6. Several observations can be made upon review of the DEGDN distribution and trend map.



- DEGDN is found in nearly all wells where RDX is present. The distribution pattern probably reflects the high solubility and mobility of DEGDN.
- The highest DEGDN concentrations were historically detected within and adjacent to the Crowd Canyon alluvium. This is close to the source area formerly located in the northeast corner of the Plant.
- Downward trends in DEGDN concentrations are evident in wells open to or in close proximity to the Crowd Canyon alluvium including, MW-1S, MW-1D, MW-2S, MW-6D, MW-7D, MW-8D, MW-11D and R-1. Downward trends are also present at Whiting, UP&L, Evans/Young, Frischknecht and R-3.
- The only upward trends in DEGDN concentrations are observed at MW-5D and MW-10D.
- DEGDN did not appear in MW-5D or Mapleton No. 1 until October 1998.
- DEGDN concentrations have never exceeded the proposed CACL of 52 µg/L at any of the monitoring wells and are not expected to do so in the future.

7.4.8 TEGDN

TEGDN production is the same as described for EGDN as a specialty nitrate. Although the exact dates of TEGDN production remain undetermined, specialty nitrates were produced at the Plant from 1976 to 1991.

7.4.8.1 *Environmental Fate and Transport*

The aqueous solubility of TEGDN is reported to be 5,500 and 6,600 mg/L at 25°C, therefore the aqueous solubility of this compound is less than these values at the ground water temperatures present in the regional aquifer (see Table 7-1). TEGDN has the second highest aqueous solubility of all of the CEMs present in the regional aquifer. The estimated Log K_{ow} values for TEGDN are the lowest for the CEMs indicating that sorption is not an important fate and transport factor for this compound. Some diffusion of TEGDN into fine grained deposits is anticipated.

7.4.8.2 *Distribution and Concentration Trends*

TEGDN data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-7. Several observations can be made upon review of the TEGDN distribution and trend map.



- TEGDN is not widely distributed and concentrations are generally lower than most other CEMs. This suggests that the production of TEGDN may have been limited when compared to EGDN and DEGDN.
- The highest TEGDN concentrations were historically detected within and adjacent to the Crowd Canyon alluvium. This is close to the known wastewater source area formerly present in the northeast corner of the Plant.
- Upward trends in TEGDN concentration are observed at only MW-1D, Evans/Young and R-1.
- There are no trends, either upward or downward, at any of the other monitoring locations.
- Detections at wells that are distal from the source area are sporadic and low.
- TEGDN has not been detected in several wells where other CEMs have been regularly detected including MW-3D, MW-5D, MW-10D, Bluth and Mapleton No. 1.
- Based on the available data, TEGDN concentrations have never exceeded the proposed CACL of 52 µg/L at any of the monitoring wells and are not expected to do so in the future.

7.4.9 TMETN

TMETN production is the same as described for EGDN as a specialty nitrate. Although the exact dates of TMETN production remain undetermined, specialty nitrates were produced at the Plant from 1976 to 1991. Plant records indicate that there were several large contracts for TMETN during the mid- to late 1980's.

7.4.9.1 Environmental Fate and Transport

The aqueous solubility of TMETN is reported to be 150 mg/L at 20°C and 516 mg/L at 19°C, therefore the aqueous solubility of this compound is less than these values at the ground water temperatures present in the regional aquifer (see Table 7-1). The Log K_{oc} estimates for TMETN are similar to those calculated for RDX indicating that sorption should be considered an important fate and transport factor for this compound. Some diffusion of TMETN into fine grained deposits is anticipated.



7.4.9.2 Distribution and Concentration Trends

TMETN data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-8. Several observations can be made upon review of the TMETN distribution and trend map.

- TMETN is found in nearly all wells where RDX is present. The distribution probably pattern reflects relatively high solubility and mobility of TMETN.
- The highest TMETN concentrations were historically detected within and adjacent to the Crowd Canyon alluvium. This is close to the source area formerly located in the northeast corner of the Plant.
- Downward trends in TMETN concentrations are observed in MW-1S, MW-6D, MW-11D, Whiting, Frischknecht, Orton-23 and R-3.
- Upward trends in TMETN concentrations are observed in MW-3D, MW-10D and Mapleton No. 1.
- TMETN has not been detected in MW-5D. TMETN first appeared in MW-10D in 1998 and did not appear in the Mapleton No. 1 well in late 1999.
- The apparent water quality boundaries observed in the RDX data are essentially the same for TMETN with the exception that TMETN is not present in MW-5D.
- Based on the available data, TMETN concentrations have never exceeded the proposed CACL of 52 µg/L at any of the monitoring wells and are not expected to do so in the future.

7.4.10 BTTN

BTTN production is the same as described for EGDN as a specialty nitrate. Although the exact dates of BTTN production remain undetermined, specialty nitrates were produced at the Plant from 1976 to 1991.

7.4.10.1 Environmental Fate and Transport

The aqueous solubility of BTTN is reported to be 800 mg/L at 20°C, therefore the aqueous solubility of this compound is less than 800 mg/L at the ground water temperatures present in the regional aquifer (see Table 7-1). The Log K_{oc} estimates for BTTN are similar to those calculated for DEGDN indicating that sorption should be considered an important fate and transport factor for this compound. Some diffusion of TMETN into fine grained deposits is anticipated.



7.4.10.2 Distribution and Concentration Trends

BTTN data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-9. Several observations can be made upon review of the BTTN distribution and trend map.

- Similar to TEGDN, BTTN is not widely distributed and concentrations are generally lower than most other CEMs. This suggests that the production of BTTN may have been limited when compared to EGDN, DEGDN and TMETN.
- The highest BTTN concentrations were historically detected within and adjacent to the Crowd Canyon alluvium. This is close to the source area formerly located in the northeast corner of the Plant.
- The only upward trends in BTTN concentrations are observed at UP&L and MW-6D, otherwise there are no trends, either upward or downward, at any of the other monitoring locations.
- Detections at wells that are distal from the source area are sporadic and low.
- BTTN has not been detected in several wells where other CEMs have been regularly detected including MW-3D, MW-5D, MW-10D, Bluth and Mapleton No. 1.
- Based on the available data, BTTN concentrations have never exceeded the proposed CACL of 52 µg/L at any of the monitoring wells and are not expected to do so in the future.

7.4.11 PETN

PETN production began in 1966 and continues to the present day. PETN nitration operations were conducted in SWMU 30 from 1966 until 1991 at which time production activities were relocated to the new PETN nitrator at SWMU 19. Discharges from the new PETN nitrator have always been treated in the wastewater treatment system and directed to the sanitary sewer. Spent acids are stored in above ground tanks and removed from the site by a third party vendor for recycling. Discharges from the old PETN operations were directed to the wastewater conveyance channel, North Impoundment and wastewater dispersion area. From late 1980 until June 1986, dilute spent nitric acid was stored in lined ponds. After the acid pond failure in 1986, all waste nitric acids have been removed from the site by a third party vendor for recycling. PETN is mixed with TNT to produce Pentolite boosters.



7.4.11.1 Environmental Fate and Transport

The aqueous solubility of PETN is less than 2.1 mg/L for the ground water temperatures present within the regional aquifer (see Table 7-1). Log K_{oc} estimates for PETN indicate that PETN has the strongest tendency to sorb to aquifer materials of all the CEMs. The sorptive characteristics of this compound may have played an important role in limiting its movement into and within the regional ground water system. PETN migration in ground water will be retarded significantly due to sorption processes relative to the other CEMs. Some diffusion of PETN into finer grained deposits is expected.

7.4.11.2 Distribution and Concentration Trends

PETN data from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-10. Several observations can be made upon review of the PETN distribution and trend map.

- PETN concentrations are generally lower than many other CEMs. PETN's low aqueous solubility and relatively high tendency to adsorb to aquifer materials probably has limited its migration in the regional aquifer.
- The highest PETN concentrations are generally detected within and adjacent to the Crowd Canyon alluvium. This is close to the source area formerly located in the northeast corner of the Plant. Interestingly, PETN has only been detected once at MW-7D and began appearing at MW-2D in September 2000, yet it has been detected more consistently and at higher concentrations in wells to the north and west (MW-1S, MW-1D, MW-6D, MW-8D, MW-11D, Whiting).
- Upward PETN concentration trends are observed at only MW-2S and MW-10D.
- The only downward trend is observed at the Bluth well.
- Detections at wells that are distal from the source area are sporadic and low.
- PETN was not detected in MW-3D until 1998, MW-10D until 1999 and Mapleton No. 1 until 2000. PETN has never been detected in MW-5D.
- Based on the available data, PETN concentrations have never exceeded the proposed CACL of 52 µg/L at any of the monitoring wells and are not expected to do so in the future.



7.4.12 Total Specialty Nitrate Esters

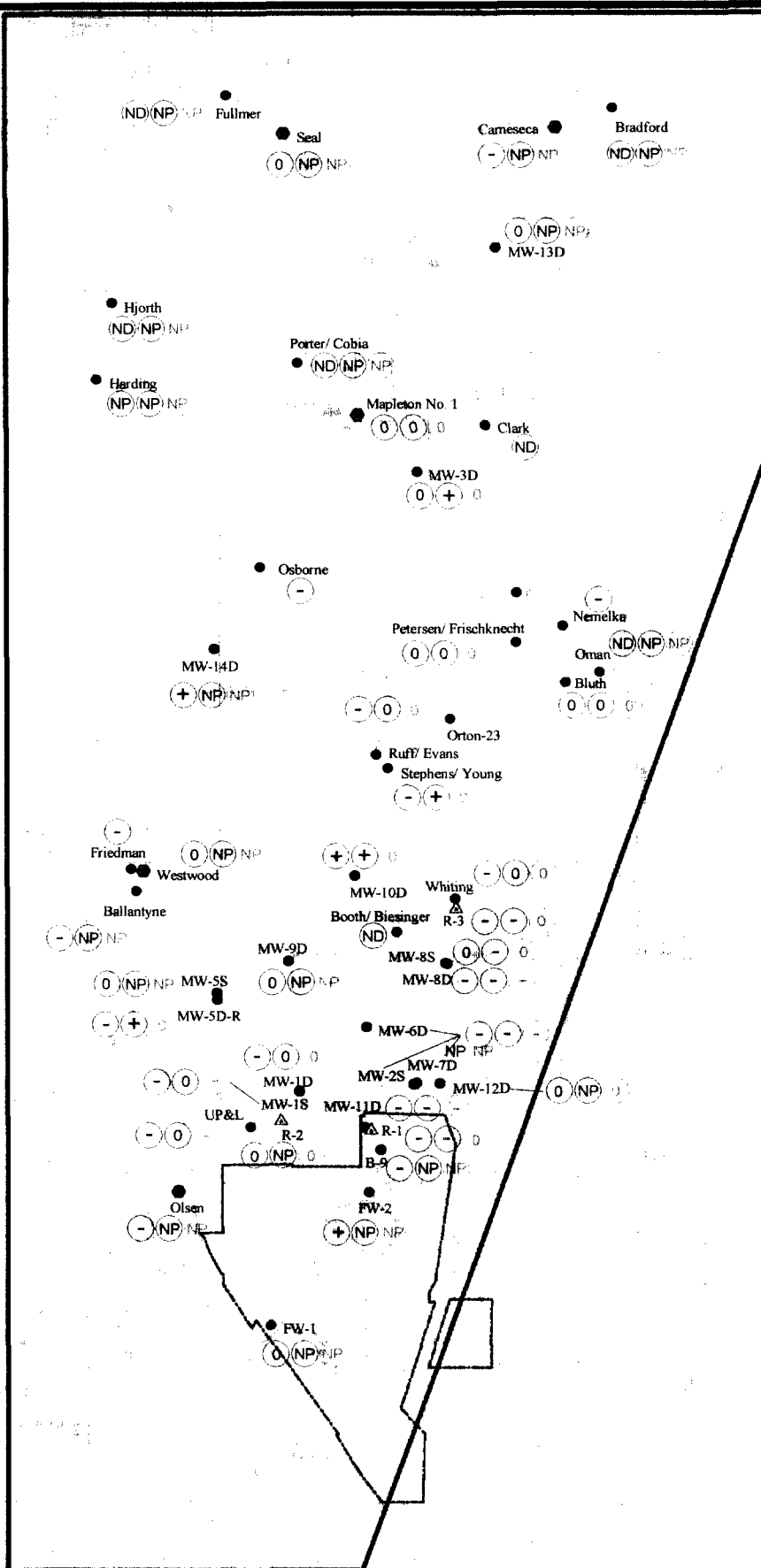
As noted in the Murphy Report, (Appendix E) the specialty nitrate esters (EGDN, DEGDN, TEGDN, TMETN, BTTN, PETN) have similar health affects. Although the compound NG is included within this category, NG has not been detected in the regional aquifer and is not part of this discussion. The proposed CACL for any single compound is the same as for the combined compounds. For this reason a distribution and trend map has been prepared for the combined specialty nitrate esters. Total specialty nitrate esters (TSNE) concentrations from the third quarter of 1995 through the fourth quarter 2001 are presented in Figure 7-11. Several observations can be made upon review of the TSNE distribution and trend map.

- As would be expected based on the distribution and trends of the individual compounds, the highest concentrations were present in wells close to the source area in the northeast corner of the Plant site.
- Downward trends in TSNE concentrations are observed at MW-1S, MW-2S, MW-6D, MW-7D, MW-8D, MW-11D, UP&L and Whiting.
- Upward TSNE concentration trends are not observed at any monitoring location.
- At the present time, the concentrations of combined specialty nitrate esters do not exceed the proposed CACL of 52 µg/L at any monitoring locations and, concentrations of combined specialty nitrate esters are not expected to exceed the proposed CACL at any time in the future. Total specialty nitrates only exceeded the proposed 52 µg/L CACL at MW-2S and MW-7D in 1995.

7.5 Expected Trends in Concentrations

Figure 7-12 characterizes concentration trends of selected COCs at several monitoring locations in the study area. The symbols at each monitoring location are indicative of concentrations trends at these locations. The statistical analyses of concentration trends were performed using the Mann-Kendall technique described in the previous section. A (+) symbol indicates a statistically significant upward concentration trend. A (-) symbol indicates a statistically significant downward concentration trend. A (o) symbol indicates that there is no statistically significant trend. A (nd) symbol indicates that the statistical significance of a potential trend could not be determined from the available data. A (np) symbol indicates that the compound is not present in the selected observation well. Samples from the Bradford, Cobia, Hjorth, Fullmer, Ballantyne, Harding and Oman wells have been analyzed for CEMs at least once, without detections. The Harding well is open to the perched Mapleton Bench ground water system and Oman is open to the bedrock aquifer. RDX, TSNE and nitrate-nitrogen have been selected to illustrate these trends. The symbols representing RDX trends are blue, TSNE are green and nitrate-nitrogen are magenta.





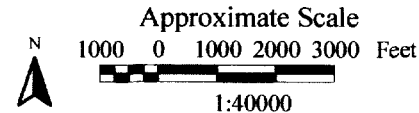
EXPLANATION

- Nitrate-Nitrogen
- RDX
- TSNE
- +** Statistically Significant Upward Trend
- Statistically Significant Downward Trend
- 0** No Statistically Significant Trend
- NP** Not Present
- ND** Trend Not Determined
- Recovery Wells
- Monitoring Wells
- Private Wells
- Municipal Wells
- EBCo Property Boundary
- Project Study Area

Oman and MW-12D are open to the Bedrock Aquifer.

Harding is open to the Perched Mapleton Bench Ground Water System.

All other wells are open to the Regional Unconsolidated Aquifer.



CHARTER OAK

4505 South Wasatch Blvd., Suite 360
Salt Lake City, Utah 84124
TEL: (801) 277-6150
FAX: (801) 277-6151

Concentration Trends Evaluated Using
The Mann-Kendall Trend Test

FIGURE 7-12

7.5.1 Nitrate-nitrogen Trends

Downward trends in nitrate-nitrogen concentration are observed in nearly all wells located proximal to the Plant site. Based on the available data, the only upward trends in nitrate-nitrogen concentrations are present at MW-10D, MW-14D and FW-2. Nitrate-nitrogen concentrations in several wells located in the approximate middle of the affected area (i.e. Orton-23, Evans/Young, and Frischknecht) have downward trends. With the exception of MW-14D, no upward trends in nitrate-nitrogen concentrations are observed in wells located at or close to the margins of the affected area (i.e. Olsen, MW-5D, Westwood, MW-3D, Mapleton No. 1).

The trends in nitrate-nitrogen concentration data indicate that concentrations are declining or stable at most locations within the affected area. Nitrate-nitrogen concentration trends at MW-14D will be the subject of ongoing evaluation. As described in Section 12.6.2 of this CAP, an additional monitoring well is being constructed in this area to evaluate ground water quality conditions.

7.5.2 RDX Trends

Downward trends in RDX concentration are observed in nearly all wells located proximal to the Plant site. No upward trends in RDX concentrations are observed in this area and MW-1S and MW-1D are the only wells in this area where there is no trend. RDX concentrations in several wells located in the approximate middle of the affected area (i.e. MW-10D, Evans/Young) have upward trends. MW-3D also has an upward RDX concentration trend. With the Exception of MW-5D, RDX is either not present or there are no upward trends in RDX concentrations in wells located at or close to the margins of the affected area (i.e. Olsen, Westwood, MW-14D, Mapleton No. 1). RDX concentrations in MW-5D were below 3 µg/L during the year 2001.

The trends in RDX concentration data indicate that concentrations are declining close to the Plant site. Increasing trends in the middle of the affected area suggests that RDX may be moving northward from the area of the Plant site. With the exception of MW-5D, RDX is either not present or there are no trends in wells along the outer margin of the affected area. The apparent movement of RDX from the Plant site toward the middle of the affected area will be the subject of ongoing evaluation.

7.5.3 TSNE Trends

Downward trends in TSNE concentration are observed in nearly all wells located proximal to the Plant site. No upward trends in TSNE concentrations are observed in this area. MW-1D is the only well in this area where there is no trend. No TSNE concentration trends are observed in wells located in the approximate middle of the affected area (i.e. MW-10D, Orton-23, Evans/Young, Frischknecht). Specialty nitrate



esters are either not present or there are no upward trends in TSNE concentrations in wells at or close to the margins of the affected area (i.e. Olsen, Westwood, MW-5D, MW-14D, Mapleton No. 1).

The trends in TSNE concentration data indicate that concentrations are declining close to the Plant site. No increasing trends are noted in wells in the middle of the affected area. TNSE are either not present or there are no trends in wells along the outer margin of the affected area.

7.5.4 Other Considerations

Production related discharges that were suspected sources of COCs and provisional COCs to ground water within the regional aquifer were eliminated at the Plant site no later than 1991 when the facility wastewater treatment system was installed. Depending upon the specific operation, potential wastewater sources were eliminated even sooner when production activities ceased. The soils data collected during the RFI indicate that the majority of CEMs in soils are limited to the upper five feet of the soil column. At the present time, perched ground water identified during the RFI is considered to be a potential source of impact to the regional aquifer and is the subject of ongoing evaluation in the RFI. Although increasing concentrations of certain CEMs are observed at selected monitoring wells, the elimination of suspected sources suggests that the future COC concentrations at any location will be no higher than the highest concentrations detected during the course of the investigation (wells closest to the source areas). Current COC concentrations are below the maximum concentrations observed in the regional aquifer. Future concentrations will remain lower than the highest historic concentrations due to the effects of dispersion, dilution, sorption and active restoration.

Diffusion of COCs into fine grained deposits is probably an important consideration for the future behavior of all COCs. The slow diffusion of COCs from ground water contained within the fine grained deposits into flowing ground water within the more permeable deposits will likely result in classic asymptotic declines in concentration. Sorption processes are likely to have a similar affect on long term concentration trends for certain CEMs. Diffusion and sorption processes are active throughout the area of impact within the regional aquifer. Both of these factors could result in concentration tailing and rebound which will be assessed as part of the recovery system performance monitoring program.

Lastly, water level changes, primarily resulting from variations in precipitation, may result in the cyclical saturation and desaturation of materials containing residual concentrations of solutes that are either sorbed to the materials or contained within immobile pore water. Periodic saturation of these materials could result in the mobilization of solutes into ground water within the regional aquifer. Concentration trends observed at MW-8S may be indicative of this process.



8.0 DATA REPORT

Substantial nitrate and CEM data have been collected throughout the hydrogeologic investigation. Copies of complete laboratory data reports were provided to DWQ in routine report submittals until January 1996. Since that time, complete laboratory data reports have not been routinely provided to DWQ; however, tabulated data summaries have been provided to DWQ on a regular basis from 1996 to the present. Scanned images of complete laboratory reports for nitrate-nitrogen, general water chemistry parameters and CEMs from 1996 through the fourth quarter of 2001 are provided in electronic format on the three CD's attached as Appendix D.

Nitrate-nitrogen and general water quality parameters are analyzed using standard laboratory methods. These analyses are performed by American West Analytical Services (AWAL). AWAL is certified by the Utah Department of Laboratory Services to perform these analyses. The method detection limit for this nitrate-nitrogen is 0.01 mg/L.

Southwest Research Institute (SwRI) provides analytical services for the analysis of CEMs. SwRI uses a modified version of SW846-8330 to analyze for CEMs. The method has been modified to enable the detection and quantification of several CEMs that are not on the standard SW846-8330 analyte list. These compounds are PETN, EGDN, DEGDN, TEGDN, TMETN and BTTN. Certified analytical standards are not available for DEGDN, TEGDN or TMETN. Non-certified standards were provided to SwRI by EBCo to facilitate analysis of these compounds. SwRI performed a method detection limit (MDL) study of the modified method in 1995. The MDL study was provided to DWQ for their review and approval. Upon review, DWQ accepted the modified method for use on this project. SwRI is certified by the Utah Department of Laboratory Services to perform SW846-8330. The method detection limits for all CEM compounds analyzed in ground water are summarized in Table 8-1. Method detection limits for these compounds are lower than the calculated RBCLs, proposed CACLs or proposed interim ground water quality goals.

This space intentionally left blank.



Table 8-1
Method Detection Limits for SW846-8330 Modified

Parameter	MDL (µg/L)
RDX	0.21
HMX	0.22
TNT	0.17
2,4-DNT	0.18
2,6-DNT	0.12
NG	0.10
EGDN	0.34
DEGDN	0.47
TEGDN	0.17
TMETN	0.23
BTTN	0.23
PETN	0.32

Overall data quality from these laboratories has been good. Some minor inconsistencies have been observed with the CEM data over time. For instance, 2,4,6-TNT and 2,6-DNT have been detected only sporadically and at a few locations. The sporadic and variable reported detections of these compounds suggest they are not present in significant quantities in off-site ground water, if at all. Specialty nitrate compounds have also displayed some irregularities over time. Certain specialty nitrate compounds have been detected at a selected well during one sampling event and not detected during another. It is not known if this is a true representation of variations in ground water quality or a function of the analytical technique. In several instances a specialty nitrate was detected in a sample and not detected in a corresponding blind duplicate sample. Because of the extensive database collected over the past five years for the CEMs, the confidence level for the identification and quantification of these compounds, with the exception of 2,4,6-TNT and 2,6-DNT, is high.

Because 2,4-DNT is likely to be present if 2,6-DNT is present, 2,4-DNT was added to the parameter list starting in the first quarter of 2001. In light of the sporadic and inconsistent detections of 2,4,6-TNT and 2,6-DNT in ground water samples, dual column confirmation is used to confirm the presence of 2,4,6-TNT, 2,4-DNT and 2,6-DNT if these compounds are preliminarily identified using the modified SW846-8330 method. The compounds 2,4,6-TNT, 2,4-DNT and 2,6-DNT were not detected in any wells open to the regional aquifer during 2001.

Based on the preliminary RFI water quality data, total and dissolved lead will be analyzed at selected wells for a period of four quarters commencing in the first quarter of 2002.

With the inclusion 2,4-DNT on the CEM analyte list and the sampling for total and dissolved lead at selected monitoring wells, there are no data gaps with respect to water quality data.



9.0 CORRECTIVE ACTION CONCENTRATION LIMITS

9.1 Proposed Corrective Action Concentration Limits

Section R317-6-6.15.F of the Utah Administrative Rules for Ground Water Quality Protection provides for the development of Corrective Action Concentration Limits (CACLS) for compounds where no State or Federal water quality criteria are available. Nitrate-nitrogen and lead are the only compounds that have a State or Federal water quality standard. The ground water quality standard established for nitrate-nitrogen by the state of Utah is 10 mg/L. The ground water quality standard for dissolved lead established by the state of Utah is 0.015 mg/L. According to section R317-6-7.15.F.2 of the Rules, DWQ will consider EPA maximum contaminant level goals, Federal Health Advisories (HA), risk-based concentration levels, standards established by other regulatory agencies or other relevant information when evaluating proposed CACLS. Federal Health Advisories are available for HMX, RDX, 2,4,6-TNT and NG. There are no published water quality standards or health advisories for 2,4-DNT, 2,6-DNT, PETN, EGDN, DEGDN, TEGDN, TMETN and BTTN.

The CAP identifies both COCs and provisional COCs. The provisional COCs have been identified as such because there have been only spurious or no detections of the constituents in the regional aquifer. In the case of lead, which is one of the provisional COCs, the Utah water quality standard is the CACL by default, so there is no need to propose a CACL. For 2,4,6-TNT, 2,4-DNT, 2,6-DNT and NG the lack of any detection of these compounds in the regional aquifer in the history of monitoring or any recent detections in the regional aquifer obviates the need to determine a CACL. Therefore, no CACLS are proposed for any of the provisional COCs.

Brian L. Murphy, Ph.D. (Dr. Murphy) has prepared a report evaluating published criteria and then developing risk based concentration levels (RBCLs) for the CEMs addressed in this CAP (Murphy Report). For completeness, published criteria and RBCLs are identified by Dr. Murphy for all COCs and provisional COCs without regard to the need to propose a CACL. The Murphy Report is included as Appendix E. In accordance with R-317-6.15.F, Dr. Murphy has also prepared an endangerment assessment. This endangerment assessment presents an analysis of several possible exposure scenarios including consumption of drinking water, consumption of vegetables and fruit irrigated with impacted water, consumption of stock that drink impacted water, consumption of fish that are raised in impacted water, dermal adsorption and inhalation.

Human health risks have been considered in developing the proposed CACLS. In estimating risks, EPA risk assessment methods have been used. These methods are intentionally conservative. The reason is that whenever there are uncertainties in estimating either exposure or chemical toxicity the EPA practice is to use a value believed to overestimate risk rather than a best estimate value.



For example, following EPA guidance, our risk-based CACLs are based on a “reasonable maximum exposure” or RME scenario. For the purposes of this assessment it has been assumed that an individual is exposed 350 days per year for 30 years even though most individuals do not live in one location for this long a period of time. It is reasonable to believe that some individuals stay in one place for thirty years, but this represents an upper estimate and most individuals typically move well before then.

The situation is similar for estimating chemical toxicity. The ability of a chemical to cause cancer is estimated from animal experiments by EPA using a statistical procedure that makes it 95% certain that the true potency value is less than the value chosen. In estimating non-cancer effects, humans are assumed to be ten times more sensitive to chemicals in general than laboratory animals, even after taking relative body weight into account. Also, non-cancer toxicity factors generally include a factor of ten to protect the most sensitive individuals in the population. Thus, the proposed risk-based CACLs are intentionally conservative in that they are based on methods designed to overestimate the true risks to the exposed population.

By EPA design, these risk estimate procedures lead to over-prediction of the actual risk. Despite these conservative assumptions and risk factors, Dr. Murphy was charged with using the most current EPA risk assessment protocols to derive the basis for the risk-based limits, which may form the basis for proposed CACLs in some cases. In the event that new scientific information concerning the COCs becomes available or any change in risk treatment of these chemicals or general revisions to EPA’s risk assessment protocols occurs, modified CACLs may be presented, adopting the new information.

This space intentionally left blank.



The CACLs for ground water within the regional unconsolidated aquifer are summarized in Table 9-1.

Table 9-1: Corrective Action Concentration Limits

Compound	CACL	Status	Source / Justification
RDX ¹	Deferred	Not Determined	See RDX discussion
HMX	400 µg/L	Proposed	HA (lifetime)
2,4,6-TNT	None	Not Applicable	Provisional COC – not detected
2,4-DNT	None	Not Applicable	Provisional COC – not detected
2,6-DNT	None	Not Applicable	Provisional COC – not detected
PETN	52 µg/L	Proposed	Dr. Murphy
NG	None	Not Applicable	Provisional COC – not detected
EGDN	52 µg/L	Proposed	Dr. Murphy
DEGDN	52 µg/L	Proposed	Dr. Murphy
TEGDN	52 µg/L	Proposed	Dr. Murphy
TMETN	52 µg/L	Proposed	Dr. Murphy
BTTN	52 µg/L	Proposed	Dr. Murphy
Total Nitrate Esters ²	52 µg/L	Proposed	Dr. Murphy
Nitrate-nitrogen ³	10 mg/L	By definition	UAC R317-6-2, Table 1
Dissolved Lead ³	0.015 mg/L	By definition	UAC R317-6-2, Table 1

¹Interim ground water quality goal of 2 ug/L is proposed until suitable data are available to establish a CACL or ACACL.

²Total nitrate esters include any combination of PETN, NG, EGDN, DEGDN, TEGDN, TMETN and BTTN

³Ground water quality protection standards have been established for nitrate-nitrogen and dissolved lead by the state of Utah (UAC R317-6-2, Table 1)

Presented below is a discussion of each COC and provisional COC and the proposal for establishing a CACL, as appropriate.

9.1.1 Nitrate-nitrogen

The Utah water quality standard for nitrate-nitrogen is 10 mg/L. As the published Utah water quality standard, 10 mg/L is the CACL and there is no need to propose one.

9.1.2 Lead

The Utah water quality standard for lead is 0.015 mg/L as dissolved lead. As the published Utah water quality standard, 0.015 mg/L dissolved lead is the CACL and there is no need to propose one.



9.1.3 NG; 2,4,6-TNT; 2,4-DNT; and, 2,6-DNT

The State of Utah does not have published water quality standards for any of these constituents.

There have been no detections of NG in the regional aquifer over the course of the history of monitoring. Therefore, NG has been identified as a provisional COC and no CACL is proposed. As described in Section 6.8 of this CAP, there have been only very few, historical and spurious detections of 2,4,6-TNT and 2,6-DNT in the regional aquifer. The compound 2,4-DNT has not been detected in the regional aquifer since it was added to analytical program in the first quarter of 2001. It is proposed that if no detections of these constituents, including NG, are recorded in four consecutive sampling events that these constituents would be removed from the provisional COC list. During 2001, none of these compounds were detected in the regional aquifer. If supported by the data, future recommendations may include the elimination of these provisional COCs from the monitoring program. Due to the lack of presence of these constituents in the regional aquifer, no CACL is proposed.

Considering the removal of the primary source of constituents to the regional aquifer, water quality is expected to generally improve over time. Therefore, consistent, future detections of these constituents are not expected. Nevertheless, if any of these constituents are detected consistently in the future, CACLs or ACACLs may need to be proposed.

9.1.4 HMX

The State of Utah does not have a published water quality standard for HMX and it is detected in the regional aquifer. Therefore, a CACL is proposed. An EPA published HA of 400 ug/L exists and is proposed as the CACL for HMX. None of the current HMX concentrations exceed this proposed standard. For the purposes of this CAP, the EPA published HA is being used; however, this does not mean that EBCo agrees that the HA is an appropriate health based standard for HMX.

9.1.5 RDX

The State of Utah does not have a published water quality standard for RDX and it has been detected in the regional aquifer. An EPA published HA of 2 ug/L exists for RDX. EBCo does not agree that this HA is based on good scientific evidence. While a risk-based concentration has been calculated for this constituent, the State of Utah has taken the position that the risk-based concentration may not be adopted as the CACL if a published HA exists. In the case of RDX, the EPA HA is not proposed as the CACL because it is founded on inadequate science. The following paragraphs are provided based on Dr. Murphy's investigation of this issue and summarize the technical shortcomings of the current EPA HA for RDX.



The current EPA HA for RDX was published in 1988. RDX health-related information is currently being reviewed in a joint DOD/EPA program whose purpose is revision of the RDX IRIS file. Most of the analysis is being conducted at the US Army Center for Health (CHPPM) at Aberdeen Proving Ground. As part of this review and revisions to the IRIS file it is likely that conclusions will be reached that are inconsistent with the present health advisory. An update on the reevaluation of the carcinogenic potential of RDX is presented in a "white paper" submitted to the Massachusetts Department of Environmental Protection (CHPPM, December 2001).

For example, the current health advisory is based in part on experiments conducted by Levine et al. (1983). These researchers determined a no adverse effect level (NOAEL) of 0.3 mg/kg/day based on prostate inflammation in male rats. This NOAEL is the basis for a Reference Dose (RfD), which in turn is the basis for the HA. According to the DOD program manager for the IRIS revision, Michael Major (410-436-7159), the prostate inflammation observations provide a very weak basis for an RfD. All of the cases of prostate inflammation were observed near the end of the study and the vast majority of cases were found in dead animals. The fact that the inflammation was suppurative suggests that bacterial infection may have been the cause. As part of the reevaluation of the HA for RDX, researchers are considering basing the RfD on a different end point: either pigment in the spleen, although that could also originate from infection; or, body weight reduction in female rats. In a study conducted by Hart et al. (1977) the NOAEL for this effect was 1.0 mg/kg/day. This change, if implemented in a straightforward manner, would increase the current HA by about a factor of 3.

The current HA also contains a modifying (reducing) factor of 10 to account for equivocal evidence of carcinogenicity. This has been EPA Office of Drinking Water policy for Group C chemicals. However, the evidence for potential RDX carcinogenicity is weaker as a result of the DOD/EPA review. The only evidence for RDX carcinogenicity was experiments in mice conducted by Lish et al. (1984). Lish et al. found a statistically significant increase in combined hepatocellular adenomas and carcinomas in female, but not male, mice. Several other experiments with rats have been negative and the chemical tests negative as a mutagen. According to Michael Major, the Lish et al. histological data has recently been reexamined by a pathologist, Dr. Parker, as part of the DOD/EPA program. His results have been reviewed and confirmed by a National Institute of Environmental Health Sciences (NIEHS) working group. A number of the growths classified as adenomas and carcinomas in the original study are now no longer classified as such. With this revision, a statistically significant increase is now seen only for one dose group, at most, below the maximum tolerated dose (MTD). The DOD/EPA program is planning an MTD study to determine if this one dose group was in fact above the MTD. Even for this one dose group, in one sex, in this one experiment, there is no statistically significant increase in "cancers," only in carcinomas and (benign) adenomas combined. Furthermore, the DOD/EPA program is also planning swine and mouse studies to examine metabolites from RDX dosed animals. They are planning to do genotoxicity testing of metabolites. If the MTD and metabolite studies result in removing



any evidence of RDX carcinogenicity, the straightforward result would be an increase in the current HA by a factor of 10.

Finally, the current HA contains a modifying (reducing) factor of 5 to account for exposure to RDX via pathways other than drinking water. This amounts to assuming that only 20% of RDX exposure comes from drinking water. It is a standard assumption made by EPA for any chemical in the absence of better information. However, as shown in Appendix E, "Development of Risk-Based Concentration Limits" to the Corrective Action Plan, reasonable assumptions lead to the conclusion that over 80% of RDX exposure would come from drinking water. Thus a straightforward application of a more realistic exposure assessment would increase the HA by about a factor of 4.

These potential changes are independent and hence multiplicative. Thus, in principle, the current HA could be increased by as much as a factor of $3 \times 10 \times 4 = 120$.

EPA has recognized the shortcomings of the RDX HA and the underlying research. In response, EPA has selected RDX for reevaluation of the toxicity information published in IRIS. EBCo proposes to wait until the new research and IRIS update is completed to propose a CACL for RDX.

Meanwhile, the ground water recovery, treatment and monitoring systems will continue to operate as protective of human health and the environment. For this interim period, EBCo proposes identifying 2 ug/L (the numerical equivalent of the HA) as the effective goal for ground water quality in the Study Area. This approach is proposed as a matter of mutual convenience to the public, the State of Utah and EBCo until such time as the scientific research concerning RDX is completed and the IRIS data-base is updated, or if technical reasons support an alternative approach. Considering the shortcomings of the current EPA HA for RDX, adopting a ground water quality goal of 2 ug/L in no way suggests that this concentration is the appropriate ultimate water quality goal.

9.1.6 Individual Nitrate Esters (PETN, EGDN, DEGDN, TEGDN, TMETN, BTTN)

The State of Utah does not have a published water quality standard for any of the individual nitrate ester constituents and all of these constituents are detected in the regional aquifer. Additionally, no published EPA HA exists for any of these constituents. Therefore, it is appropriate to propose risk-based concentration limits (RBCLs) for each of these constituents. The RBCLs calculated by Dr. Murphy are proposed as the CACLs for the individual nitrate esters. The proposed CACL of 52 ug/L pertains to PETN, EGDN, DEGDN, TEGDN, TMETN and BTTN.

9.1.7 Nitrate Esters as a Group (PETN, EGDN, DEGDN, TEGDN, TMETN, BTTN, NG)

Due to common physiological effects and structural similarities the nitrate esters have also been evaluated as a group. As a conservative measure, it is proposed that any



combination of nitrate esters should also be assigned an RBCL. The State of Utah does not have a published water quality standard for the nitrate esters as a group and there is no published EPA HA for this grouping of constituents. Therefore, it is appropriate to propose an RBCL for this group of constituents. Although NG has not been detected in the regional aquifer, it is considered to be in the classification of nitrate esters and the health effects are similar to the other six nitrate esters. Therefore, as a conservative measure, NG concentrations will be counted in the total nitrate ester concentration as compared to the proposed CACL. The proposed CACL for nitrate esters as a group (e.g. total nitrate esters) is 52 ug/L.

9.2 Future CACLs and ACACLs

The CACLs proposed in this revised CAP are based on the best information currently available. In the future, EBCo reserves the right, as appropriate, to petition the State of Utah to modify an existing CACL, to add CACLs for constituents where none are currently proposed and to propose the adoption of ACACLs.



10.0 CORRECTIVE ACTION

In accordance with Section R317-6-6.15.D.2 of the Utah Administrative Rules for Ground Water Quality Protection, a description of the corrective measures to address off-site ground water impacts is presented herein. As noted earlier in this document, a corrective action has already been implemented to address off-site impacts to ground water in the study area. The installed corrective action is more comprehensive than what was originally proposed in the Interim Measures Work Plan (Owens Western, 1996b). A description of the completed corrective measures is presented herein.

10.1 Ground Water Extraction Facilities

10.1.1 Extraction Wells

The ground water extraction system consists of five ground water extraction wells operating in the Spanish Fork Study area. Two wells (Mapleton No. 1 and Orton-23) are existing, high volume municipal/irrigation wells adapted for this corrective action and three extraction wells were installed specifically for the ground water remediation project. The term "recovery well" is used interchangeably with "extraction well" in this document. Extraction well locations are presented in Figure 10-1. Well logs and construction information for the extraction wells are provided in Appendix F. Each of these wells essentially operates continuously except for operation and maintenance or data collection interruptions.

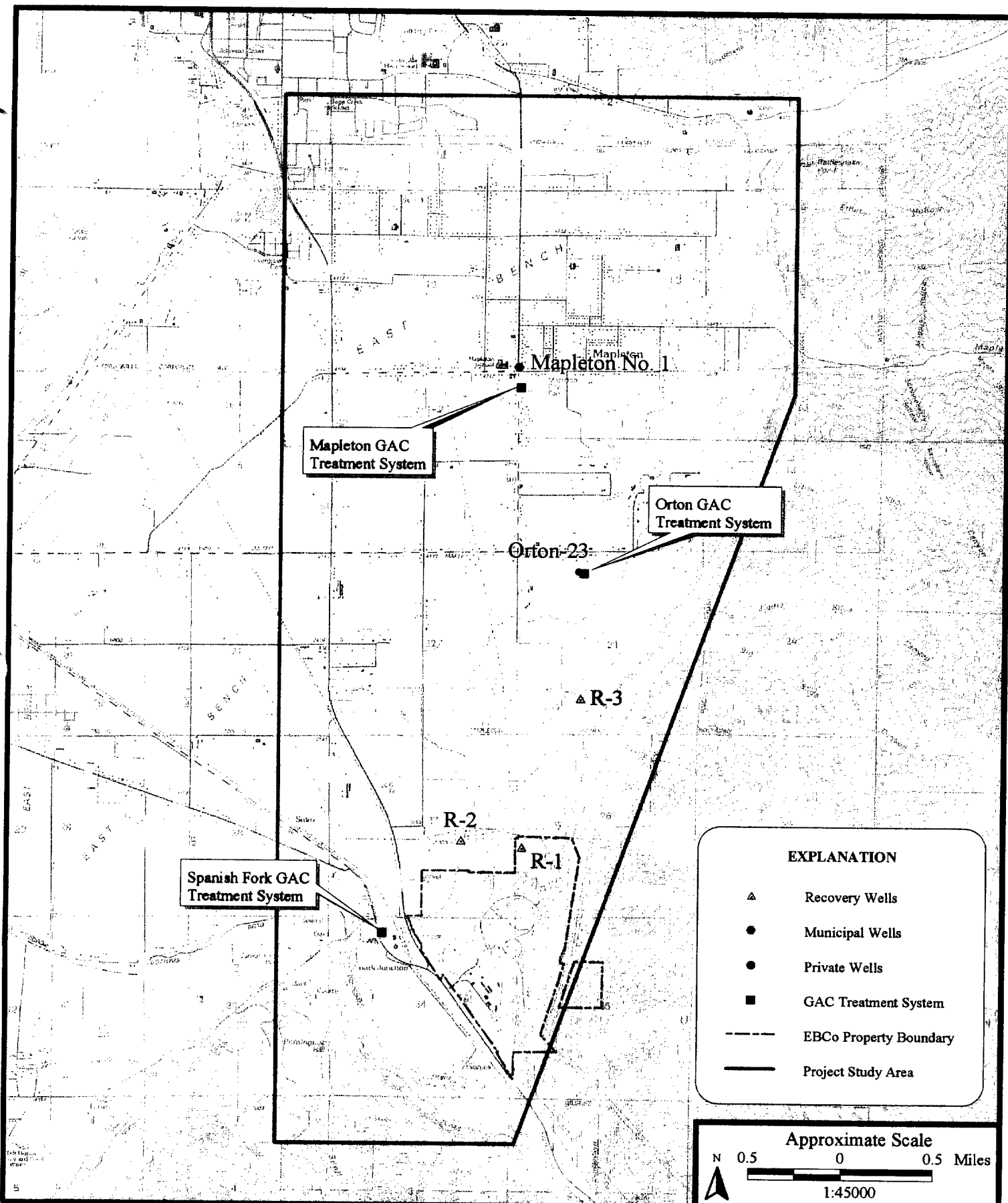
10.1.1.1 Mapleton No. 1 Well

The Mapleton No. 1 well is located in downtown Mapleton at the intersection of Maple and Main streets. It is a sixteen-inch steel cased well completed at a depth of 500 feet below grade. The well intake area consists of perforated steel casing from 383 to 494 feet below grade. The Mapleton No. 1 well is capable of producing between 1000 and 1100 gpm of water with its current motor and pump system. Mapleton No. 1 continues to withdraw ground water even when its use is not required by Mapleton City to meet water supply needs.

10.1.1.2 Orton-23 Well

The Orton-23 well is located in Mapleton at approximately 750 East and 1600 South. It is a twelve-inch steel cased well completed at a depth of 345 feet below grade. The well intake area consists of perforated steel casing in three discrete depth zones: The upper perforated section is from 190 to 200 below grade; the middle perforated section is from 252 to 262 below grade and the lower perforated section is from 290 to 325 below grade.





CHARTER OAK
 ENVIRONMENTAL SERVICES, INC.
 4505 South Wasatch Blvd., Suite 360
 Salt Lake City, Utah 84124
 TEL: (801) 277-6150
 FAX: (801) 277-6151

Extraction Well and GAC Treatment System Location Map

FIGURE 10-1

The Orton-23 well is capable of producing between 800 and 1100 gpm of water with its current motor and pump system. Additional information and details about the Orton-23 well can be found in the R-1, R-2 and Orton-23 Well Construction and Pump Test Report (Charter Oak, 1998c) and the Orton-23 Recovery Well Packer Test Report (Charter Oak, 2001d). The pump in this well has been equipped with a variable frequency drive to adjust flow rate, as necessary, to avoid excessive drawdown in this well. The Orton-23 well continues to withdraw ground water even when its use is not required for irrigation.

10.1.1.3 Recovery Well R-1

Recovery well R-1 is located in the northeast corner of the EBCo site. It is a ten-inch steel cased well completed at a depth of 450 feet below grade. The well intake area consists of two sections of continuous wrap, stainless steel well screen from 285 to 371 feet and 386 to 436 feet below grade. The R-1 well is capable of producing between 300 and 500 gpm of water. Additional information and details about the R-1 well can be found in the R-1, R-2 and Orton-23 Well Construction and Pump Test Report (Charter Oak, 1998c).

10.1.1.4 Recovery Well R-2

Recovery well R-2 is located just east of the Pacificorp electrical substation in Spanish Fork. It is a ten-inch steel cased well completed at a depth of 280 feet below grade. The well intake area consists of continuous wrap, stainless steel well screen from 231.5 to 266.5 feet below grade. The R-2 well is capable of producing between 30 and 50 gpm of water. Additional information and details about the R-2 well can be found in the R-1, R-2 and Orton-23 Well Construction and Pump Test Report (Charter Oak, 1998c).

10.1.1.5 Recovery Well R-3

Recovery well R-3 is located on privately owned property in south Mapleton. It is a ten-inch steel cased well completed at a depth of 380 feet below grade. The well intake area consists of continuous wrap, stainless steel well screen from 280 to 370 feet below grade. The R-3 well is capable of producing between 40 and 60 gpm of water. Additional information and details about the R-3 well can be found in the Recovery Well R-3 Well Construction and Pump Test Report (Owens Western, 1997).

10.1.2 Ground Water Treatment

10.1.2.1 Nitrate-nitrogen

For the current uses of recovered ground water no treatment is necessary for the removal of nitrates from the recovery system discharge. Low nitrate-nitrogen concentrations present in the recovery wells (approximately 3 to 11 mg/L) are suitable for use in



pressurized irrigation systems or for permitted discharges to surface waters (Hobble Creek and Spanish Fork River). Nitrate-nitrogen concentrations in the Mapleton No.1 well have varied from approximately 2 to 8 mg/L since 1989 and are below the 10 mg/L MCL for nitrate-nitrogen. The use of water from the Mapleton No. 1 well is unrestricted based on nitrate-nitrogen concentrations.

Several options for nitrate treatment were examined as part of the ongoing assessment of remediation alternatives throughout this project. These nitrate treatment alternatives included reverse osmosis, ion exchange, land application, dilution (surface water discharge or in-line mixing) and sanitary sewer discharge. Reverse osmosis and ion exchange proved to be technically impractical due to the inability to effectively manage relatively large volumes of high ionic strength regeneration discharge. The Spanish Fork Publicly owned Treatment Works (POTW) is presently unable to handle the projected volume and strength of discharges from nitrate treatment. Lined storage and evaporation impoundments for the management of nitrate treatment discharges are impractical due to land requirements (see An Evaluation of Wastewater Management Alternatives, CEE 1997). Direct discharge to the sanitary sewer is also not feasible because of capacity issues at the Spanish Fork POTW and in sanitary sewer conveyance piping. Disposal costs for this alternative also proved to be prohibitive. Ground water resource use in the municipal secondary water systems and surface water discharge are being used to manage nitrate-bearing ground water collected from ground water extraction facilities. Nitrate treatment has become unnecessary due to the general decline in nitrate concentrations in the study area and use of the extracted ground water as a resource for irrigation purposes.

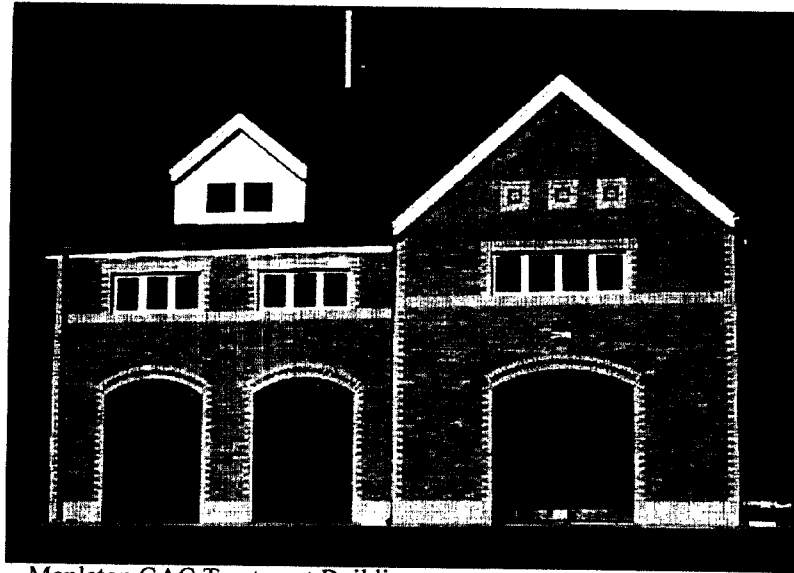
10.1.2.2 Constituents of Energetic Materials (CEMs)

Granulated activated carbon (GAC) adsorption is the treatment method selected to remove CEMs from the recovered ground water. The feasibility of GAC treatment was assessed along with UV/Oxidation as part of the Off-site Contamination Feasibility Study (Owens Western 1995b). GAC was selected because it was a proven and cost effective treatment method for low concentrations of CEMs in ground water.

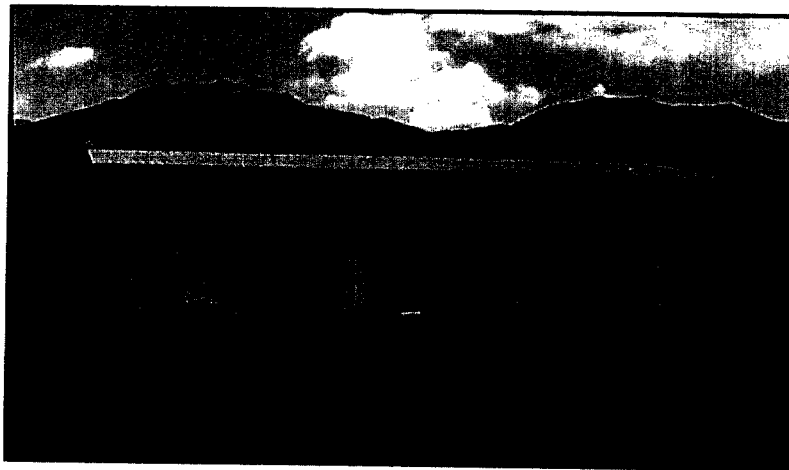
Three nearly identical GAC treatment facilities have been installed in the project area: Mapleton GAC, Orton GAC and Spanish Fork GAC. The three systems were fabricated and installed by Northwest Carbon (now US Filter) of Red Bluff, California. Their locations are presented in Figure 10-1. Images of each of the GAC treatment buildings are provided in Figure 10-2. Figure 10-3 presents some images of a GAC treatment system that are representative of the three treatment systems installed. The three GAC treatment systems were constructed after receiving design approval from DWQ. Approval of the Mapleton GAC treatment system was also obtained from the Utah Division of Drinking Water (DDW). DDW approval allows the use of the Mapleton GAC treatment system to treat water prior to possible introduction into Mapleton's potable water supply system. The DWQ and DDW approval letters are provided in Appendix G.



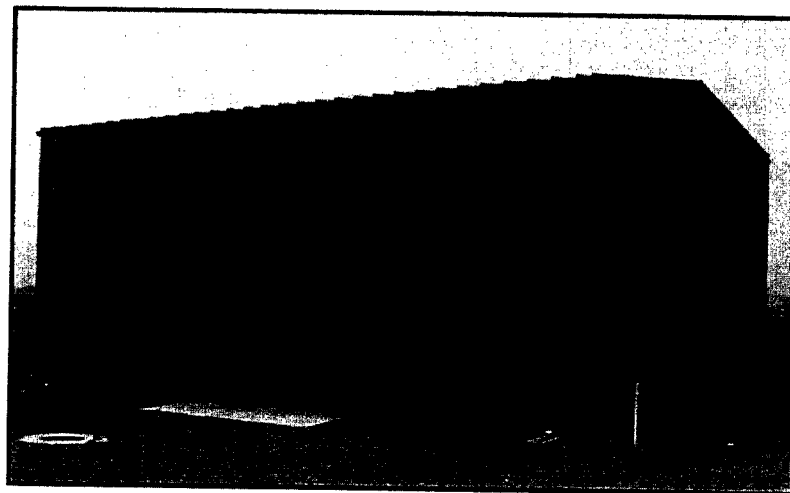
Figure 10-2: GAC Treatment Buildings



Mapleton GAC Treatment Building

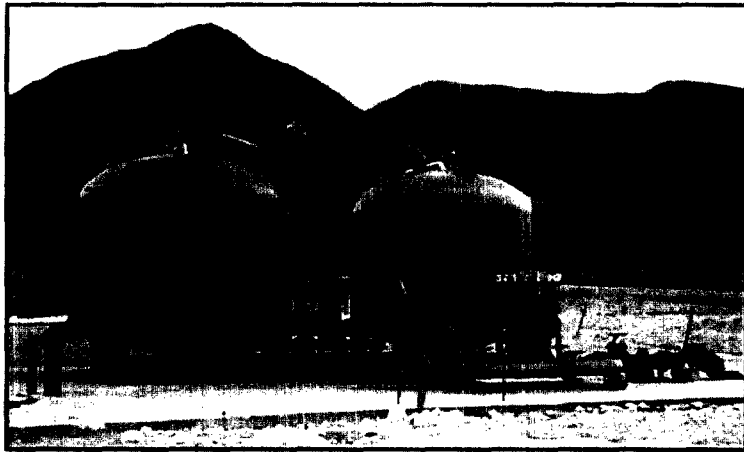


Orton GAC Treatment Building

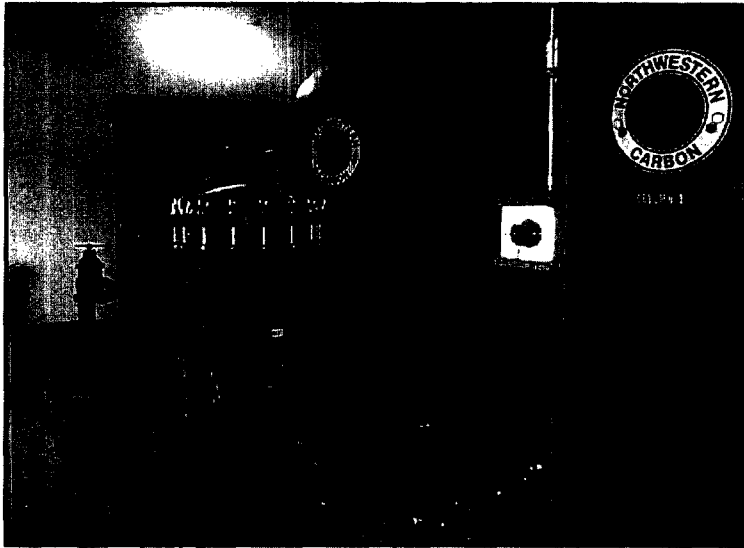


Spanish Fork GAC Treatment Building

Figure 10-3: Typical GAC Treatment System



Two 20,000 lb treatment vessels



Pre-filter, backflow preventer and treatment vessels



Carbon performance sample ports and flow control manifold

Spent carbon from the GAC treatment facilities does not exhibit the reactivity characteristic and does not contain listed hazardous wastes. Therefore, the spent GAC can be reactivated and placed back into service. Virgin, reactivated and spent carbon is shipped and managed exclusively by US Filter. The spent carbon remains segregated and tracked at the US Filter carbon reactivation facility, ensuring that only carbon used at the Spanish Fork project GAC treatment systems is regenerated and returned to any one of the three GAC treatment systems.

Because water from the Mapleton GAC treatment system may, at Mapleton's discretion, be connected to the potable water supply system, the use of reactivated carbon in the Mapleton GAC treatment system must be in compliance with DDW's "Policy Statement on Use of Reactivated Carbon For Drinking Water Treatment" (Policy Statement). The Policy Statement requires that the carbon regeneration facility (RF) meet the following criteria:

1. The RF does not handle carbon which would be classified as a hazardous waste.
2. The RF's procedures for transport of spent carbon both prior to and after reactivation must demonstrate the transport vehicles and equipment are only used for nonhazardous materials and should be properly cleaned between pickups.
3. The RF must have adequate procedures for maintaining adequate separation of spent carbon being reactivated from the time of removal through the reactivation process and during storage until reinstallation.
4. The RF must have an adequate quality control program that must include: (a) reactivation to Food Chemical Code Standards; (b) continuous activity testing to insure quality uniformity throughout the reactivation process; (c) re-screening to original size and similar particle distribution; (d) segregated storage until reinstallation; and, (e) minimal turn around time to insure the water system's ability to meet supply requirements.
5. The RF must provide an annual statement indicating any changes that they may have instituted.
6. The contract with the RF must specify that the system's own carbon (all three GAC treatment facilities combined) must be returned to it, that it must be segregated at all times, and that only virgin carbon can be used to make up the original volume.
7. The RF must evaluate calcium levels in the influent to determine if "hard" water (>75 mg/L CaCO_3) is present. Elevated hardness can result in calcium adsorption onto the carbon and subsequent leaching of aluminum and possibly iron after reactivation. In accordance with the Policy Statement, EBCo has elected to the run the first bed volume to waste after reinstallation to address this issue.



8. The RF must perform third party iodine testing of the virgin carbon to establish baseline adsorptive capacity for that carbon. Subsequent iodine testing must be performed after each reactivation to ensure proper reactivation of the carbon.

DDW has approved the use of US Filter for the reactivation and supply of carbon to the Mapleton GAC treatment system. A copy of the DDW approval letter is provided in Appendix H.

Approximately 8,000 – 16,000 gallons of water discharges are produced during each carbon exchange event. This water is derived from the slurry transfer process used to remove and install the carbon in each treatment vessel and from a short duration backwashing operation that expands and settles the carbon once placed in the treatment vessel. The water is discharged to the Spanish Fork POTW via either Mapleton or Spanish Fork sewerage conveyance lines. The water discharge is sampled for CEMs, nitrate and total suspended solids (TSS) and Charter Oak maintains these data for future reference. Details regarding the operation and maintenance of the GAC treatment facilities are provided in two documents: the Mapleton GAC O&M Manual and the Spanish Fork GAC O&M Manual (Charter Oak, revised April 2002). Details regarding carbon exchange activities are provided therein.

Each treatment vessel is equipped for sampling at five locations (influent, 25%, 50%, 75% and effluent (100%)). Additionally, the influent and effluent samples can be collected from each two-vessel system. Carbon usage and performance at each treatment facility is assessed through a monthly sampling and analysis program. At least four samples are collected from each of the treatment vessels (influent, effluent, and two internal samples). The data are reviewed monthly to determine carbon loading at each treatment facility. Carbon is generally replaced when carbon in the lead vessel is nearly fully loaded and some breakthrough to the lag column is identified. Typically, carbon in the lead treatment vessels in both the Mapleton and Orton GAC treatment systems is replaced just prior to, or near the beginning of the irrigation season that begins in April of each year. However, this schedule is continually evaluated based upon monthly carbon performance data and if the carbon in these systems exhibits a slower loading rate this annual replacement frequency may be modified. Carbon in the Spanish Fork GAC system has demonstrated a slower loading rate; therefore, the carbon replacement at the Spanish Fork GAC system is scheduled based upon the monthly carbon performance data.

10.1.3 Treated Ground Water Management

Once impacted ground water is extracted and treated to remove CEMs, the treated water must be managed. As noted previously, there are a limited number of feasible options available for the handling of the high volume (approximately 2,500 gpm) of treated water produced by ground water recovery operations. Water from the recovery system is managed in two ways.



10.1.3.1 Pressurized Irrigation

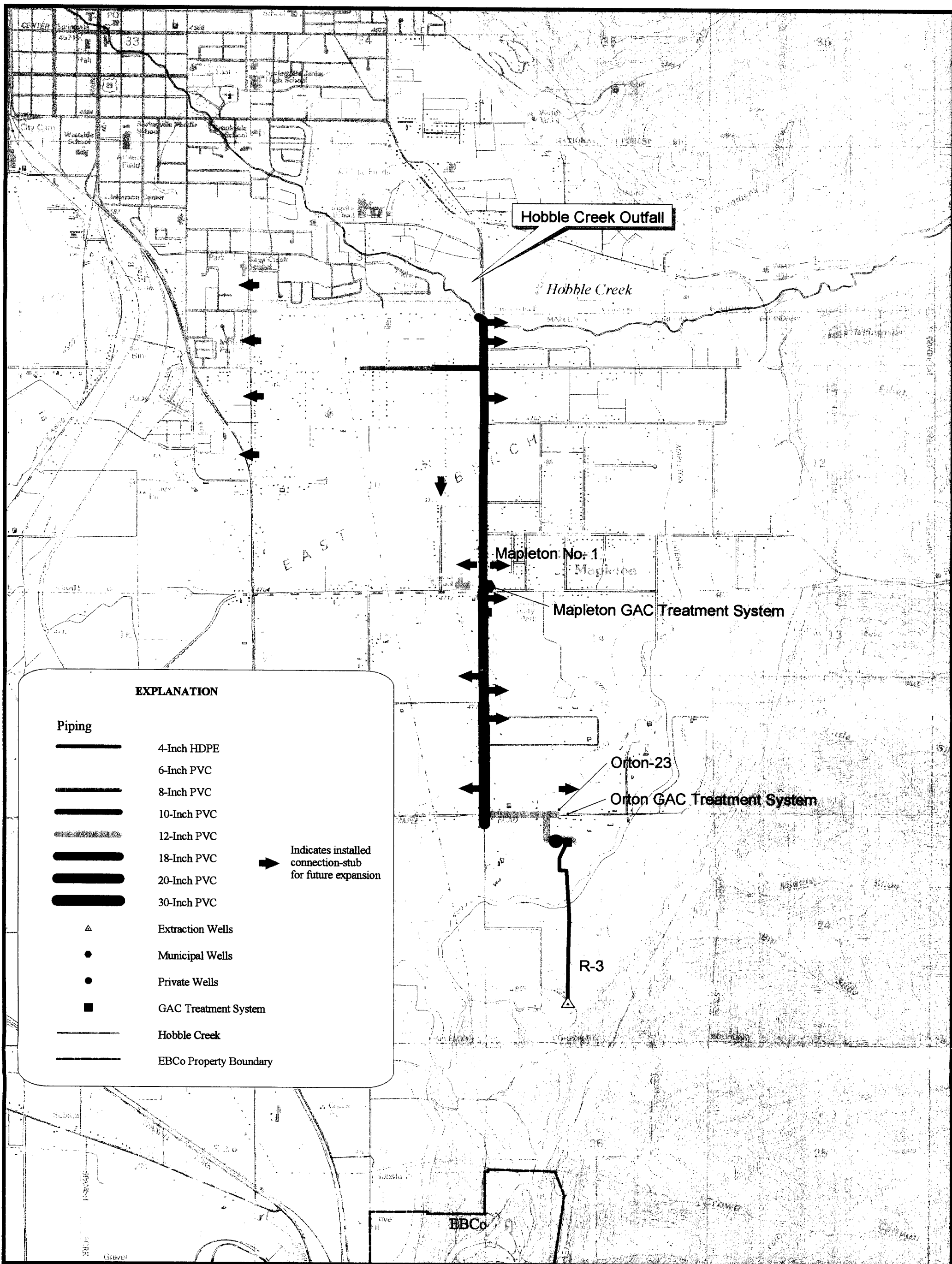
Restoration of the ground water resource to beneficial use was an important consideration for EBCo. Use of recovered and treated ground water in municipal pressurized irrigation (PI) systems facilitates the continued beneficial use of the ground water resource and helps to minimize the use of municipal ground water resources for irrigation purposes.

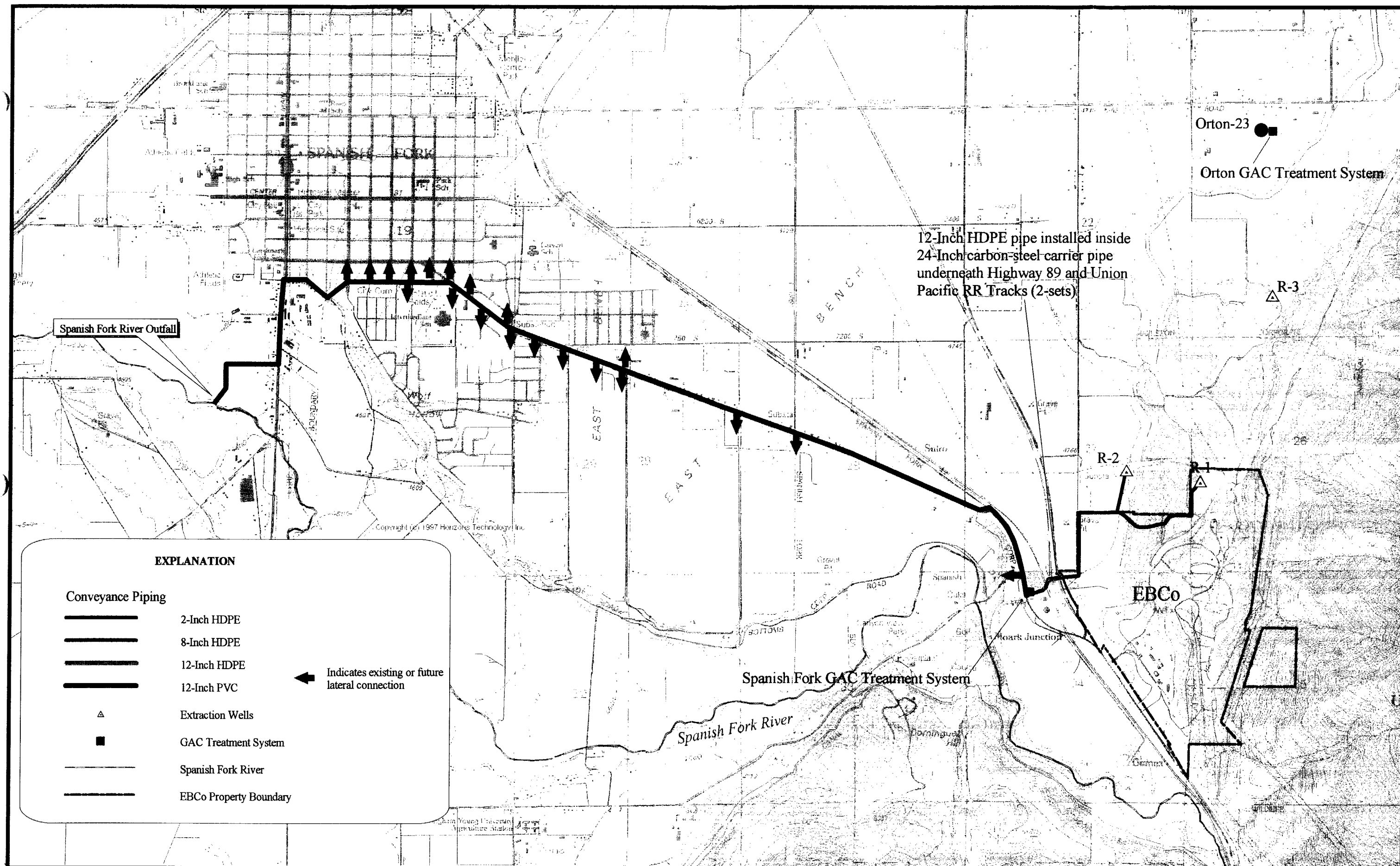
During the irrigation season (April 1 through October 31) the treated water is made available to the cities of Spanish Fork and Mapleton for use in their PI systems. These systems are used exclusively for the irrigation of municipal and privately owned properties in those two communities. EBCo installed extensive conveyance and irrigation lateral piping in Mapleton to enable use of recovered water in Mapleton's PI system. Components of the Mapleton PI system were oversized to accommodate anticipated future irrigation water supply from the Central Utah Project (CUP). EBCo also provided financial assistance to Spanish Fork for the installation of the "Canyon Road Pipeline" that facilitated the connection of the R-1 and R-2 recovery wells to Spanish Fork's existing PI system. The general layout and sizes of the installed conveyance piping for Mapleton and Spanish Fork is illustrated in Figures 10-4a and 10-4b, respectively. Excess water that cannot be fully utilized in the PI systems during the irrigation season is discharged to Hobbie Creek and the Spanish Fork River in accordance with the UPDES permits referenced above. During the non-irrigation season (November 1 through April 31) treated water is directed to the surface water streams via the installed conveyance piping. Over 40,000 linear feet of conveyance piping and irrigation laterals have been installed as part of the corrective action. Spanish Fork utilizes nearly all the water from the R-1 and R-2 recovery wells (approximately 400 gpm) during the irrigation season. During the 2001 irrigation season, Mapleton used about forty percent of the approximately 1,000 gpm produced from the Mapleton No. 1 well. As Mapleton's PI system becomes more fully developed, it is anticipated that all the recovered water will be utilized in the PI system during the irrigation season.

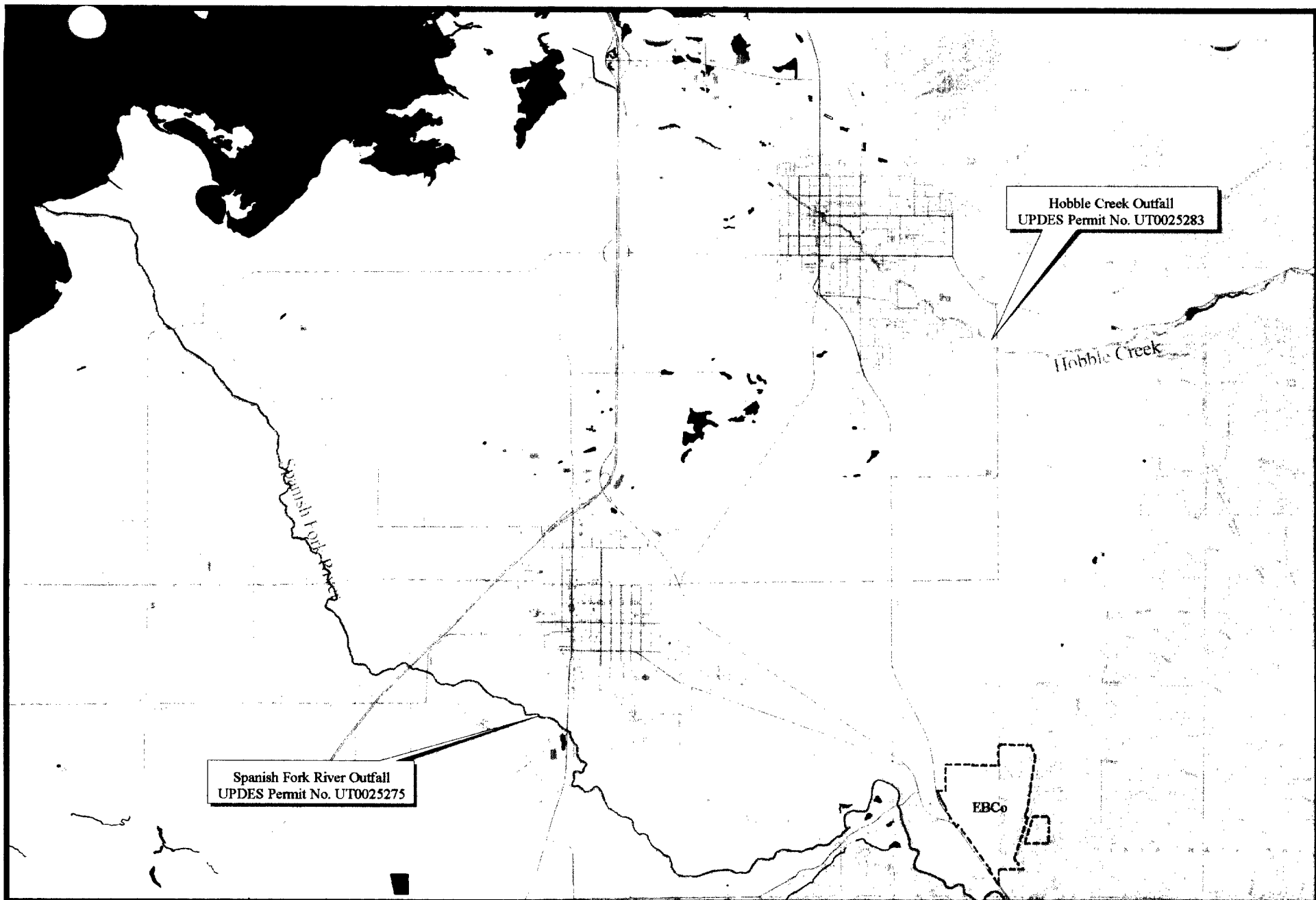
10.1.3.2 Surface Water Discharge


Treated waters are discharged to both the Spanish Fork River and Hobbie Creek at the locations illustrated in Figure 10-5. Treated water from recovery wells R-1 and R-2 is directed to the Spanish Fork River and treated water from the Mapleton No. 1, Orton-23 and R-3 wells is directed to Hobbie Creek. EBCo applied for and DWQ granted Utah Pollution Discharge Elimination Program (UPDES) permits for the two discharges. UPDES Permit No. UT0025283 addresses the Hobbie Creek discharge and UPDES Permit No. UT0025275 addresses discharges to the Spanish Fork River. The UPDES permits specify monitoring and reporting requirements and effluent limits for dissolved oxygen (D.O.), pH, nitrate-nitrogen and RDX. The permits also specify requirements for quarterly two species, chronic and acute whole effluent toxicity (WET) testing. In accordance with permit requirements, EBCo prepares monthly discharge monitoring reports (DMR), which report monitoring results to DWQ. DMR's are available for









CHARTER OAK 
 4505 South Wasatch Blvd., Suite 360
 Salt Lake City, Utah 84124
 TEL: (801) 277-6150
 FAX: (801) 277-6151

Approximate Scale
 N 0.5 0 0.5 1 1.5 Miles
 1:80000

Surface Water Discharge Points

FIGURE 10-5

review at DWQ. Copies of the discharge permits and correspondence detailing subsequent permit modifications are attached in Appendix I. Specifics regarding the permit requirements and effluent limits for each of these discharges can be found in the attached documents.

10.2 Natural Attenuation

According to the EPA Directive entitled *Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action and Underground Storage Tank Sites* (EPA, 1999), "monitored natural attenuation" as a remedial strategy is the reliance on natural physical, chemical or biological processes that act to reduce the mass, toxicity, concentration, mobility, volume or concentration of constituents on soil or in ground water. These *in-situ* natural attenuation processes include biodegradation, dispersion, dilution, sorption, volatilization, radioactive decay and chemical or biological stabilization, transformation or destruction of constituents. EPA prefers natural attenuation processes that degrade or destroy constituents; however, this does not preclude the recognition of other, non-destructive natural attenuation processes as part of a natural attenuation program. For this CAP, natural attenuation supplements the active ground water recovery and treatment system that has been installed in the study area. This conforms to EPA's stated expectation that natural attenuation will be most appropriate when used in conjunction with other remediation measures such as source control and ground water extraction. EPA also notes that natural attenuation is most appropriate for sites where the impacted regions are no longer increasing in size or are shrinking.

For the purposes of this CAP, emphasis is placed on those natural attenuation mechanisms that are relevant to solutes present in the regional aquifer system. Solute behavior and concentration trends indicate that natural attenuation processes are active both within and outside of the zone of active restoration. As discussed in Section 7, photolysis, hydrolysis and volatilization are not considered to be relevant natural attenuation mechanisms for the regional aquifer. Based on extensive literature review and a site-specific evaluation of postulated breakdown products for RDX, the available data do not support biodegradation as a mechanism that is measurable or observable in the regional aquifer. Nitrate is considered to be a conservative solute and does not sorb to aquifer materials. While all CEMs have a low propensity to sorb to aquifer materials relative to many other organic compounds, certain CEMs will sorb to aquifer materials resulting in the temporary or permanent removal of certain CEMs from ground water and the retardation of solute migration. Some studies have demonstrated that TNT may form irreversible covalent bonds with soil particles resulting in the permanent removal of TNT from impacted water. Therefore, sorption may be a viable natural attenuation process for TNT in some environments. However, it is likely that sorption for most of the CEMs is reversible and sorbed CEMs will desorb back into the flowing ground water as a result of equilibrium partitioning.

Based on the foregoing and consideration of other data and observations, dilution and dispersion are the most important natural attenuation processes active in the regional aquifer. According to Wiedemeier et. al. (1996), dilution results in the reduction of



solute concentrations and an apparent decrease in the total mass of solutes in a system due to the introduction of additional water to the system. Dilution does not destroy or transform constituents. Dilution in the study area is caused by ground water recharge, where unaffected water (recharge) mixes with the impacted water in the regional aquifer resulting in dilution. In the study area dilution occurs primarily within the foothills recharge area along the eastern margins of the study area. Evidence of dilution is most apparent in the regional aquifer within and adjacent to the Crowd Canyon alluvium. Rapid declines in nitrate-nitrogen concentrations coincide with the termination of surface discharges in 1991 and an increase in water levels from 1994 through 1998. The rise in water levels reflects increased recharge to higher than average amounts of precipitation over this time period. Because other potential natural attenuation mechanisms (i.e. biodegradation, sorption) are not applicable to nitrate, dilution is considered the primary natural attenuation mechanism. CEMs also exhibit declining concentration trends in this area. However, concentration declines for certain CEMs, such as RDX and HMX, are less rapid than that of nitrate-nitrogen, possibly reflecting desorption of these compounds from aquifer materials. Dilution may also be an important natural attenuation process in the extreme northern and southern margins of the study area where the regional aquifer is recharged by Hobble Creek and the Spanish Fork River.

Dispersion, defined in Section 5.2.1.2 of this report, is the other important natural attenuation mechanism acting in the regional aquifer. Dispersion of a solute into unaffected portions of the regional aquifer allows the three-dimensional mixing of solutes with unaffected ground water, thereby reducing solute concentrations at the edges of the affected area. In the absence of other natural attenuation processes, the low solute concentrations and/or lack of increasing solute concentration trends along the approximate margins of affected area (i.e. Westwood, Mapleton No. 1, MW-5D) may be indicative of natural attenuation due to dispersion, in these areas. Solute concentration trends in the area of MW-14D, where nitrate-nitrogen is present at about 5 mg/L but no CEMs have been detected, will be the subject of ongoing evaluation and assessment.

Production-related discharges at the Plant have been directed to a wastewater treatment system and sanitary sewer since 1991. The notable declines in solute concentrations in proximity to the former wastewater management area located in the northeast corner of the Plant are consistent with the termination of discharges in this area. The extent to which impacted on-site soils or perched ground water may act as continuing sources of solute concentrations to the regional aquifer is the subject of the ongoing RFI. Because of the lack of continuing discharges to the ground, site soils are not being loaded with additional constituent mass. Considering the foregoing, although some constituents have exhibited upward concentration trends at selected locations, overall concentrations within the regional aquifer are expected to decline over time.

10.3 Water Quality Monitoring

Water quality sampling of selected private, municipal and monitoring wells is performed on a quarterly schedule. Recovery wells are sampled on a monthly frequency. Water levels in recovery wells are measured weekly and water levels from selected observation



wells are measured at least monthly. The present water quality monitoring program is in general accordance with the monitoring program proposed in the CAP (Charter Oak, 2001b) and approved by DWQ in letters dated November 6, 2001 and January 18, 2002. The primary purpose of this sampling is to assess overall water quality within the regional aquifer in study area. These data are used to assess solute distribution and long-term trends in nitrate-nitrogen and CEM concentrations. These analyses are used to determine if the affected area is expanding beyond present limits, to determine if concentrations are increasing or declining at various locations and/or to assess the effectiveness of the corrective action. If these data indicate that sampling of additional existing wells or the installation of new monitoring wells is warranted to track water quality changes, additional work will be recommended to DWQ.

10.4 Institutional Controls

10.4.1 Private and Municipal Well Reporting

Upon the receipt and review of water quality sampling results, EBCo provides water quality data summaries to Mapleton, Spanish Fork and individual private well owners whose wells are part of the sampling program. Copies of these letters are provided to DWQ and the Utah County Department of Health. Complete data summaries for each quarterly sampling event are also reported to DWQ.

10.4.2 Well Owner Notification

In 1997, water rights records available for the study area were reviewed. Based on the water rights information and discussions with Mapleton City representatives, well owners were identified and well owner information was compiled in a database. EBCo sent letters, via certified mail, to well owners informing them of known or suspected water quality conditions in their well, along with a brief description of planned corrective action activities. Copies of these letters were supplied to the DWQ and the Utah County Department of Health.

10.4.3 Annual Review of Water Use

Water rights records for the Spanish Fork study area are reviewed annually. These records are compared with an existing water rights database to identify potential new well owners in the study area. If new well owners are identified in areas of known ground water impacts EBCo will notify them of known or suspected water quality conditions in their well.



10.4.4 DWR Limitations on Water Right Transfers

In accordance with the Utah/Goshen Valley Ground Water Management Plan (DWR, 1995) a "restricted area" in the study area has been established by DWR to address ground water impacts in the area. This "restricted area" was established to "...protect public health and reduce the migration of these contaminants..." According to the management plan, DWR should not grant new change applications, which propose to transfer water rights into this area. The boundaries of this area are illustrated in Figure 10-6.

10.4.5 Other Activities

10.4.5.1 Building Permit Requirements

An ordinance established in Mapleton City's municipal code does not allow issuance of building permits without a municipal water connection. This policy ensures that new homes/businesses will not rely on a private well for potable water.

10.4.5.2 Public Meetings

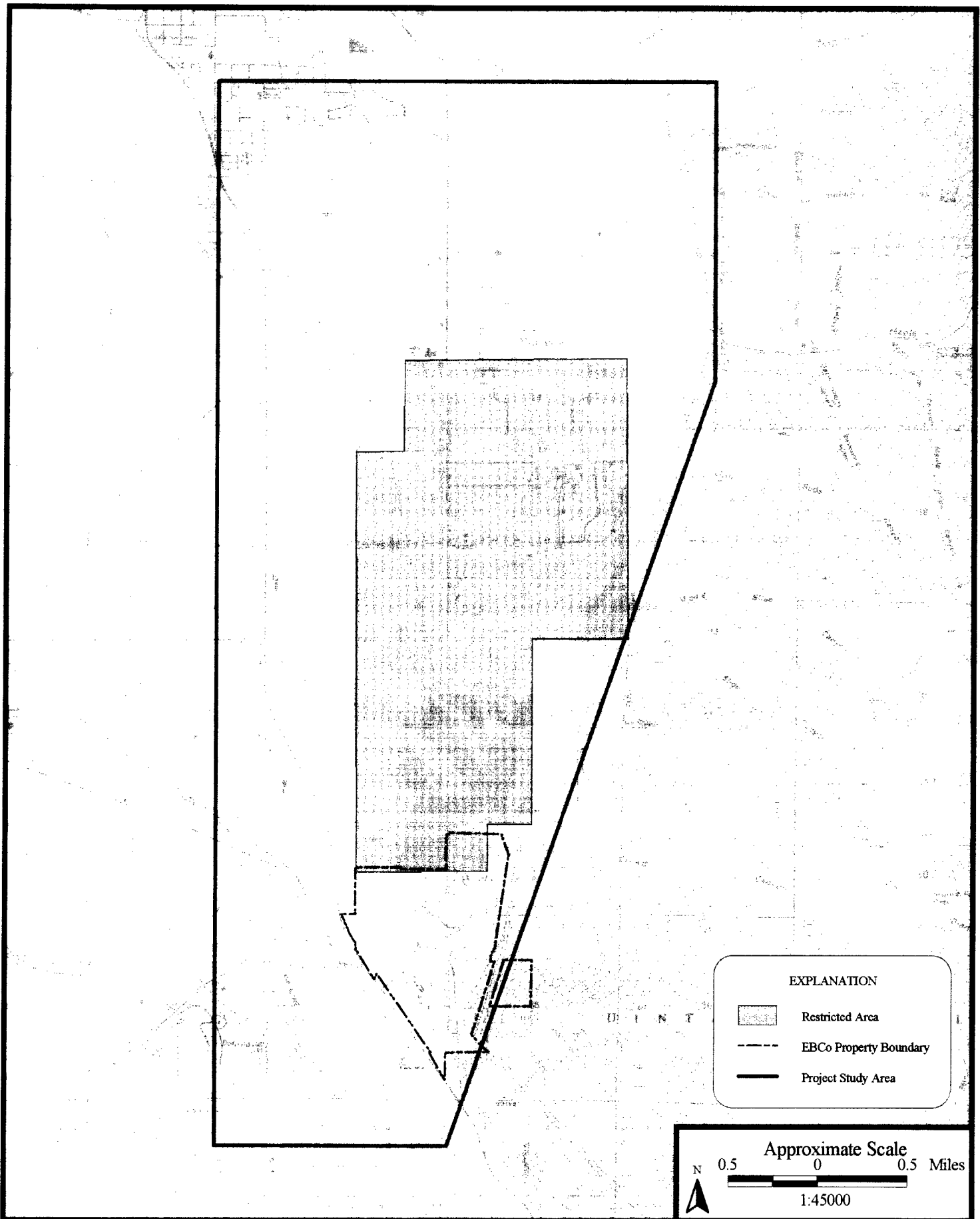
EBCo and DEQ meet at least annually (more frequently if deemed necessary) to discuss project activities. Representatives of Mapleton and Spanish Fork are typically invited to attend these meetings. Additionally, if local municipal representatives request additional information or meetings with EBCo representatives, EBCo generally accommodates these requests. DWQ and EBCo representatives have participated in several meetings in Mapleton including City Council meetings, meetings with the Mayor, a public hearing and public availability sessions. There will be a public notice to allow public input on this CAP.


10.5 On-site Controls

10.5.1 RFI Investigation and Corrective Measures

In accordance with the Resource Conservation and Recovery Act (RCRA) an extensive sampling and analysis program as part of a RCRA Facility Investigation (RFI) is substantially complete. The purpose of the RFI is to identify on-site soil, surface water and ground water conditions to enable evaluation of potential risks to human health and the environment. Upon completion of the RFI, an RFI summary report containing sampling results, a human health risk assessment, an ecological risk assessment and of particular importance to this CAP, an evaluation of the potential for continuing impact to on- and off-site ground water. Based on the findings of the RFI, a Corrective Measures Study (CMS) will be prepared that will evaluate and propose remediation activities to





CHARTER OAK 
 4505 South Wasatch Blvd., Suite 360
 Salt Lake City, Utah 84124
 TEL: (801) 277-6150
 FAX: (801) 277-6151

**Restricted Area Established by the Utah
 Department of Natural Resources,
 Division of Water Rights**

FIGURE 10-6

) eliminate or control potential threats to human health and the environment, to the extent they exist. Once the DSHW accepts the CMS, corrective measures will be implemented. Upon completion of the corrective measures, potential sources of contamination will be removed to approved levels and/or on-site controls will be in place that will be protective of human health and the environment. This work is being performed in accordance with a consent order between EBCo and DSHW and is being coordinated with off-site activities addressed in this CAP. The on-site corrective measures will be handled under the RCRA program and are not addressed further in this CAP.

10.6 Additional Considerations

While ground water extraction, treatment and water delivery systems are now fully operational, numerous technical and administrative issues had to be addressed before the corrective actions could be implemented. Some of the more significant issues are identified herein.

10.6.1 Water Rights

) Utah water law requires that any withdrawal of ground water must be in accordance with an approved water right. Because underground water rights in Southern Utah Valley have been over-allocated (there are more paper water rights than actual water available), the DWR has closed the Southern Utah Valley to new appropriations. There are no provisions in Utah Water Law to allow for the extraction of water for ground water remediation purposes; therefore EBCo is subject to the same water rights restrictions as all other water users. EBCo had to acquire or otherwise negotiate the use of water rights to facilitate the discharge or beneficial use of extracted ground water. EBCo worked closely with DWR, Spanish Fork and Mapleton for approximately two years to secure sufficient water rights to allow operation of the ground water extraction facilities.

10.6.2 Property Access and/or Approvals

With the exception of recovery well R-1 and some conveyance piping related to that well, the entire recovery system, including recovery wells, conveyance piping, GAC treatment facilities, meter vaults and discharge structures are located on private, state or municipally owned property. Because of this, access agreements and easements had to be negotiated and/or permits had to be filed in order to facilitate the construction of the recovery system.

) Access agreements and or construction permits had to be obtained from the following entities:



Entity	Recovery System Element
Bureau of Reclamation	Mapleton Lateral pipe crossing (two)
Mapleton Irrigation Board	Mapleton Lateral pipe crossing (two)
Mapleton City	Pipeline construction, well usage, GAC treatment system building
Spanish Fork City	Pipeline construction, GAC treatment system building
Pacificorp	Well construction, pipeline construction
Property Reserve Inc.	Pipeline construction
Southern Pacific Railroad	Railroad pipe crossing
Utah Department of Transportation	Pipeline construction, highway pipe crossing
Three private landowners	Well construction, well usage, pipeline construction, GAC treatment system building

10.6.3 Water Management

The installed recovery system is capable of producing up to 2,500 gpm (3.6 mgpd) of treated ground water. Management of this substantial volume of water requires special consideration. CEE and Owens Western performed an evaluation of water management alternatives to determine the most technically feasible and cost-effective water management alternatives. Several criteria were used to evaluate the alternatives including technical practicability, time to implement, impacts on human health and the environment, community benefit, land access, water rights, cost, regulatory consideration and public acceptance. The results of that evaluation are presented in the report entitled "Evaluation of Water Management Alternatives, Spanish Fork, Utah (CEE, 1997). Water management alternatives examined included:

- Mapleton PI System
- Spanish Fork PI System
- Surface Water Discharge
- Sewer Discharge
- Wetlands Treatment
- Phytoremediation
- Land Application
- Subsurface Reinjection
- Shallow Infiltration
- Deep Well Injection
- Seasonal Pumping Analysis



As a result of this evaluation, a combination of the Mapleton PI system, the Spanish Fork PI system and surface water discharge was selected. The technical practicability, time to implement, community benefit and public acceptance criteria overwhelmingly supported the selected alternatives.



11.0 ASSESSMENT OF CORRECTIVE ACTION

In accordance with sections R317-6-6.15.E.(1-5) of the Utah Administrative Rules for Ground Water Quality Protection (DWQ, 1995) a preliminary assessment of the corrective action is presented herein. Consistent with DWQ objectives the proposed corrective action is protective of human health and the environment and will produce a permanent effect of meeting appropriate concentration limits.

One objective of this corrective action is to ensure protection of human health and the environment. Here, the principal affected environmental media is ground water present within the regional aquifer. Because impacted ground water within the regional aquifer does not discharge to springs, drains, or surface water bodies, these potential receptors are not of concern. The corrective action, consisting of ground water extraction, GAC treatment and natural attenuation reduces the spread of solutes, removes solute mass and reduces solute concentrations. This is protective of the affected environmental media, regional ground water.

The extraction and treatment of ground water is also protective of human health. Concentrations of CEMs in the treated ground water are typically below analytical detection limits and have remained below the proposed CACLs or in the case of RDX, the proposed interim ground water quality goal of 2 µg/L. Nitrate-nitrogen concentrations in the combined recovery well flows are below 10 mg/L. The high quality of the treated water effectively restores the beneficial use of this water. The proposed CACLs, which are calculated based on current EPA methodologies or existing health advisories and the proposed interim ground water quality goal for RDX are protective of human health. The institutional controls that are in place are also protective of human health by minimizing the potential for inadvertent exposure to impacted ground water.

Another objective of the corrective action is to lower constituent concentrations within the regional aquifer to acceptable levels (proposed CACLs or proposed interim ground water quality goal for RDX). The elimination of discharges to the ground, which are suspected to have been the primary sources at the Plant site, has resulted in the substantial improvement of water quality within the regional aquifer at locations proximal to the Plant. Because these sources have been eliminated, the potential for future impacts to the regional ground water from these sources has also been essentially eliminated. The potential for residual concentrations in soil to represent a continuing threat to ground water is being evaluated in the RFI, and will be addressed as appropriate in that context. Based on observed declining solute trends proximal to the facility, residual soil concentrations do not currently appear to represent a significant continuing threat to ground water within the regional unconsolidated aquifer. Extraction and treatment of ground water and natural attenuation should result in a permanent lowering of constituent concentrations in the regional aquifer. Due to the heterogeneous nature of the regional aquifer and the affects of sorption and diffusion, the time frame for reaching the water quality objectives remains uncertain. This will be the subject of ongoing monitoring and evaluation.



11.1 Ground Water Extraction

A review of recovery system performance data collected since the start of ground water extraction and treatment operations through the end of calendar year 2001 is presented in this section.

11.1.1 Extraction Well Operation and Discharge Rates

The following figures (Figures 11-1 through 11-5) present discharge data for each extraction well. These charts are useful for evaluating changes in discharge rates over time. Changes in discharge rate may be caused by either increasing pumping heads (due to water level declines) or by the manual lowering of discharge rates. These charts also show the pumping history of the recovery wells for comparison with the water level changes observed in the hydrographs presented in the following section. The intermittent spikes in recorded flow rate generally reflect periodic inaccuracies in the Sensaphone automated logging equipment used to record the flows.

11.1.1.1 Mapleton No. 1

Figure 11-1 presents the discharge history for the Mapleton No. 1 well. Mapleton No. 1 discharge remained relatively constant at a rate of between approximately 1,000 and 1,100 gpm until August 2001. Since August 2001 the discharge rate has been between approximately 950 and 1,000 gpm. The higher flow rates from August to mid-September 1998 reflect the use of the original pump and motor system, which had a higher discharge rate than anticipated. A smaller pump and motor system was installed in September 1998 and has been in use since. Based on available data, the long-term sustainable flow rate from this well is expected to be between 950 and 1,100 gpm.

11.1.1.2 Orton-23

Figure 11-2 presents the discharge history for Orton-23. The discharge rate in this well has declined from approximately 1,200 gpm in March 1999 to approximately 800 gpm in December 2001. The decrease in discharge rate reflects manual flow controls to maintain a suitable pumping water level, as well as an apparent decline in discharge rate as pumping water levels have decreased over time. The Orton-23 was shut down or operated at a reduced flow rate from March 2001 until August 2001 to enable packer testing and modifications to the well and pump system. As a result of this testing, the pump has been equipped with a variable frequency drive and automatic level control to avoid excessive drawdown in this well. The long-term sustainable flow rate for this well is expected to be between 800 and 900 gpm; however, future changes in flow rate may be required in response to changing water level conditions.



11.1.1.3 R-1

Figure 11-3 presents the discharge history for R-1. The discharge rate in this well has declined from approximately 450 gpm in July 1999 to approximately 350 gpm in December 2001 with most of this decrease occurring since October 2000. The decrease in discharge rate reflects manual flow controls to maintain a suitable pumping water level, as water levels have declined in the regional aquifer. The long-term sustainable flow rate for this well is expected to be between 300 and 400 gpm; however, future changes in flow rate may be required in response to changing water level conditions.

11.1.1.4 R-2

Figure 11-4 presents the discharge history for R-2. The discharge rate in this well has declined from approximately 50 gpm in July 1999 to approximately 30 gpm in December 2001. The decrease in discharge rate reflects manual flow controls to maintain a suitable pumping water level, as water levels have declined in the regional aquifer. The long-term sustainable flow rate for this well is expected to be between 30 and 40 gpm; however, future changes in flow rate may be required in response to changing water level conditions.

11.1.1.5 R-3

Figure 11-5 presents the discharge history for R-3. The discharge rate in this well has declined from approximately 120 gpm in May 1999 to approximately 65 gpm in November 2000. The decrease in discharge rate reflects manual flow controls to maintain a suitable pumping water level, as water levels have declined in the regional aquifer. R-3 was shut down in November 2000 to assess the effect that pumping of the Orton-23 well has on water level trends in this area. It also was determined that the pump in R-3 was damaged and it has been replaced with a lower capacity pump with a projected maximum flow rate of between 40 and 70 gpm. Since being returned to service in September 2001, the R-3 well has been pumping at a rate of between approximately 50 and 70 gpm. The long-term sustainable flow rate for this well is expected to be between 40 and 60 gpm; however, future changes in flow rate may be required in response to changing water level conditions.

11.1.2 Water Level Response

Hydrographs for the extraction wells and selected observation wells help illustrate the influence that the recovery wells have on water levels within the regional aquifer. These hydrographs present data from the approximate start of extraction well operations through the end of calendar year 2001. In the future, water levels will rise and fall in response to changing pumping conditions in the regional aquifer and varying amounts of



Figure 11-1: Mapleton No. 1 Flow Log

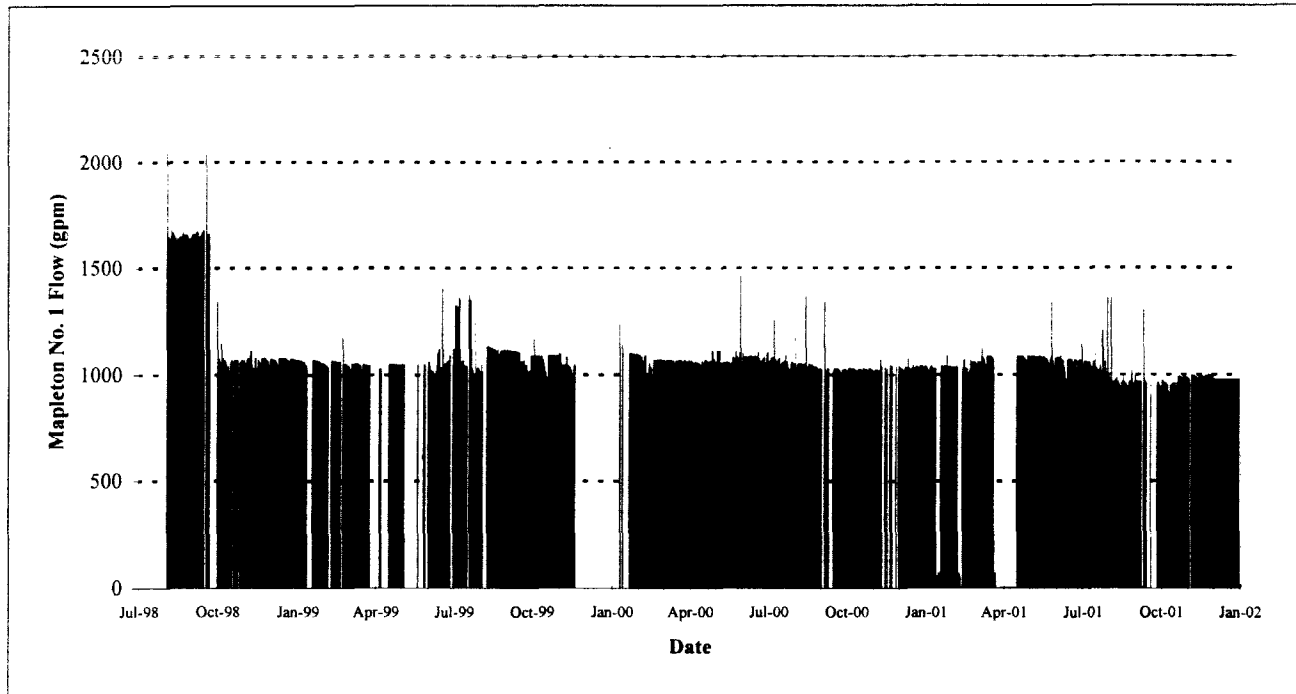


Figure 11-2: Orton-23 Flow Log

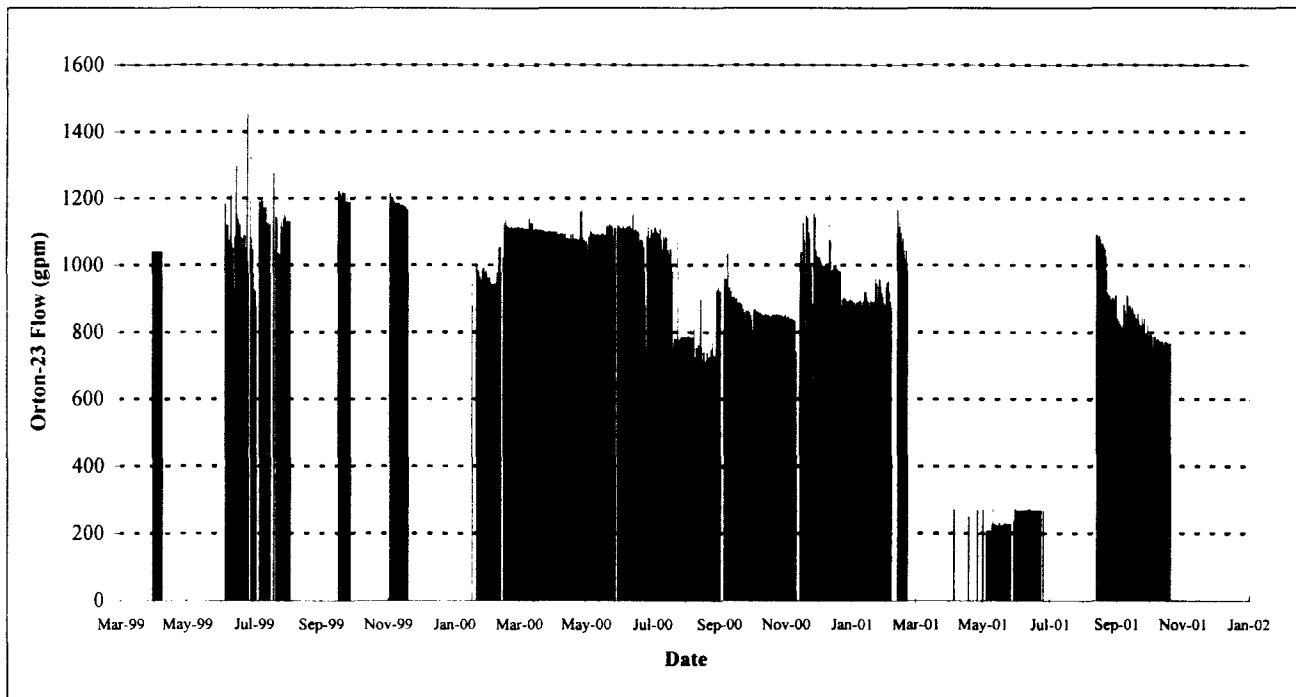


Figure 11-3: R-1 Flow Log

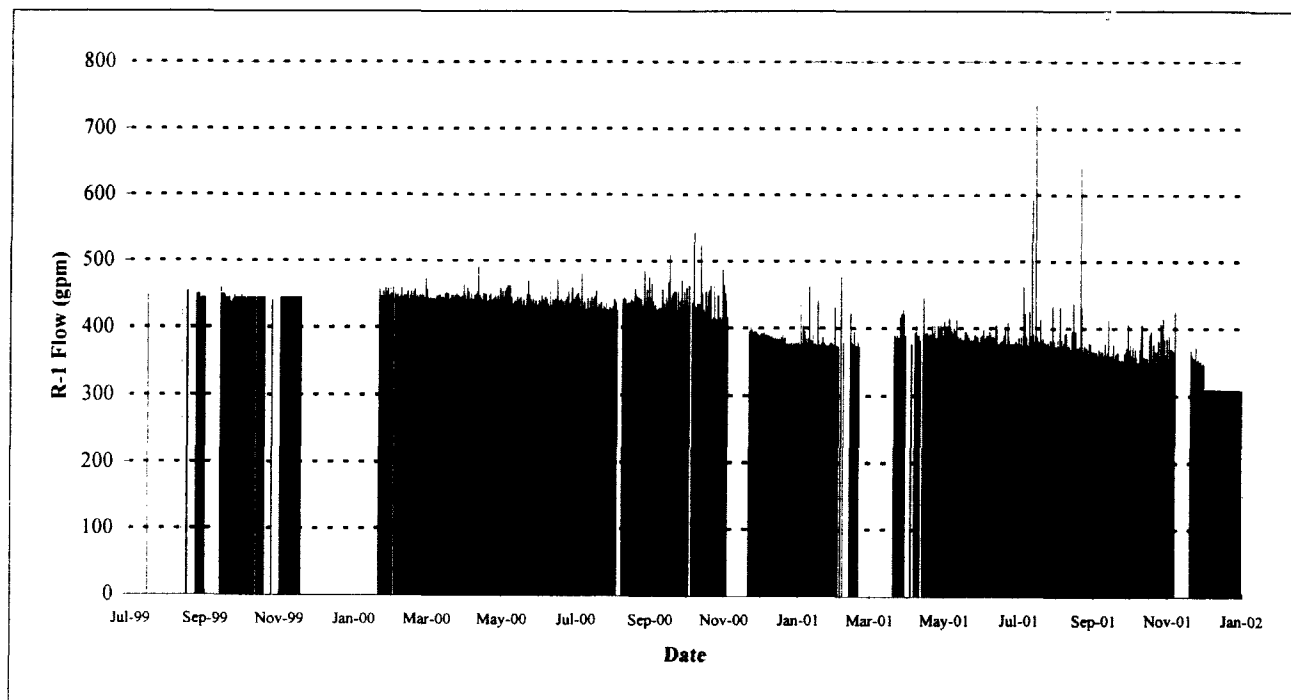


Figure 11-4: R-2 Flow Log

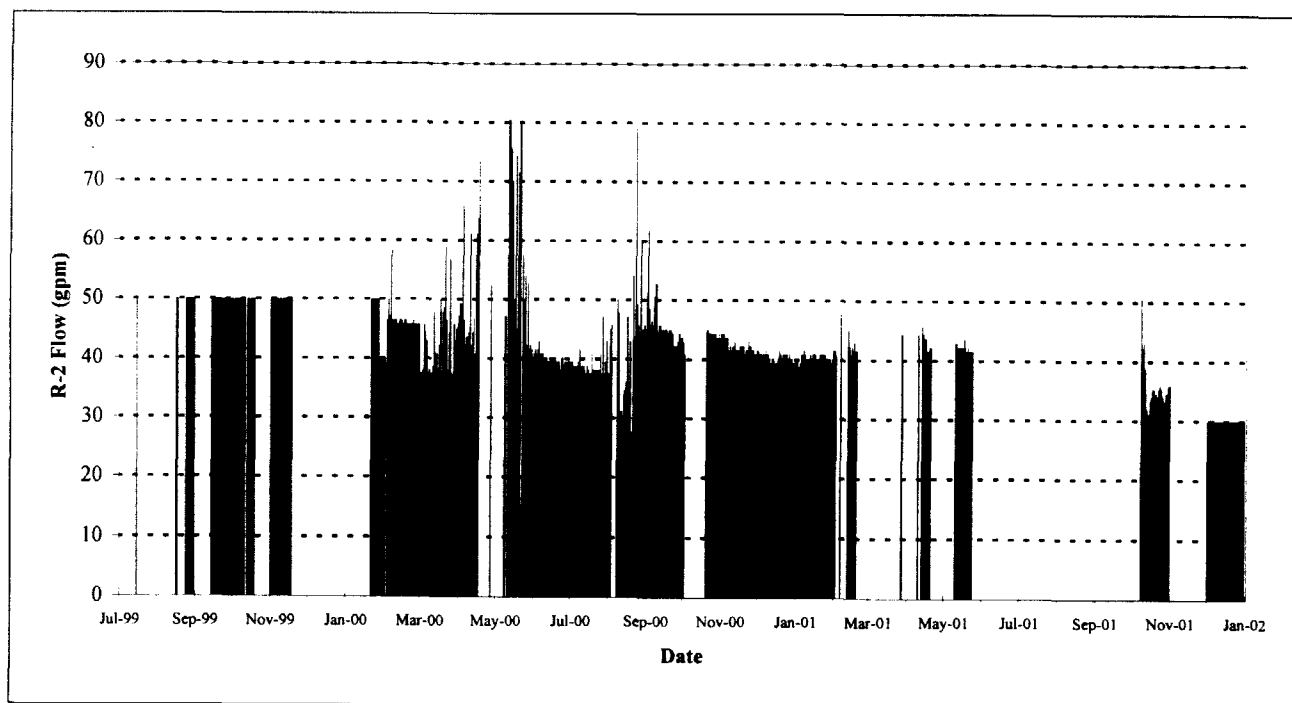
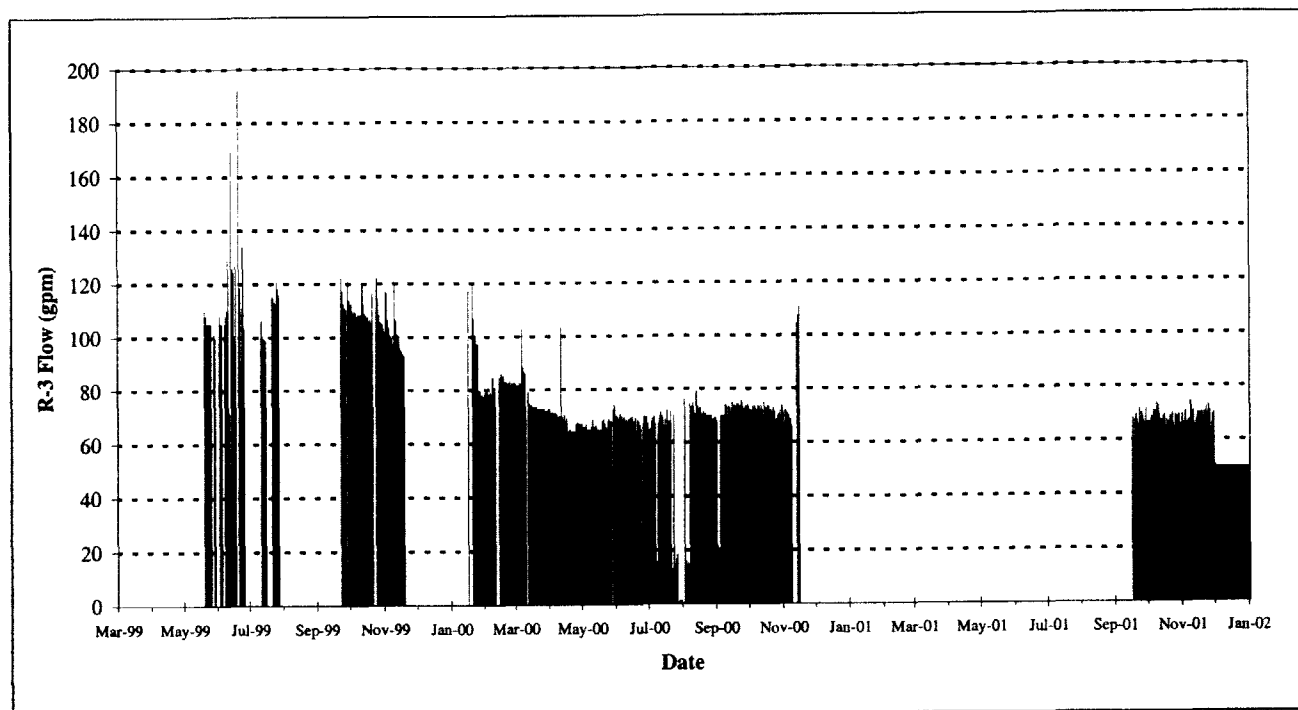


Figure 11-5: R-3 Flow Log



ground water recharge due to both seasonal and long-term changes in the amount of precipitation.

11.1.2.1 Mapleton No. 1

Figure 11-6 presents hydrographs for the Mapleton No. 1 well and observation wells MW-3D, MW-13D and MW-14D.

An immediate drawdown of about five feet is observed in Mapleton No. 1 at the start of pumping. Approximately two feet of drawdown is observed at MW-3D in response to pumping of Mapleton No. 1. No obvious immediate water level responses are observed at MW-13D and MW-14D in response to pumping of Mapleton No. 1. An overall maximum regional water level decline of approximately thirty feet is apparent in all of the wells reflecting the drop in water levels due to below average recharge and the pumping of the No. 1 well and other high capacity wells including Orton-23, Seal and Carneseca. The rise in water levels from November 1999 through January 2000 largely reflects the temporary cessation of recovery system operations during this time period. The rise in water levels observed from September 2000 through April 2000 probably reflects reduced pumping in the regional aquifer as the irrigation season water demands were reduced. A similar rise in water levels is observed from November 2001 through December 2001. The relatively small water level responses observed at Mapleton No. 1 and the observation wells reflects the relatively high transmissivity of deposits in this area.

11.1.2.2 Orton-23

Figure 11-7 presents hydrographs for Orton-23 and observation wells MW-10D, Bluth and Young. An immediate drawdown of about twenty feet is observed in Orton-23 at the start of pumping. Immediate water level responses are observed at each of the observation wells with declines of about six feet at Bluth, five feet at Young and three feet at MW-10D. An overall maximum regional water level decline of approximately thirty feet is apparent in all of the wells reflecting the drop in water levels due to below average recharge and the pumping of the Orton-23 well. The rise in water levels from November 1999 through January 2000 largely reflects the temporary cessation of recovery system operations during this time period. The rise in water levels from July 2000 through August 2000 reflects a 30% reduction in the discharge rate of Orton-23 over this time period. The rise in water levels observed from September 2000 through April 2000 in Mapleton No. 1 and nearby observation wells is not as obvious in these hydrographs. The leveling of water levels during the September 2000 through February 2001 time period probably reflects a combination of reduced pumping in the regional aquifer and lower discharge rates from Orton-23. The rise in water levels between March and August 2001 reflects the limited use of the Orton-23 well during this time period.



11.1.2.3 R-1

Figure 11-8 presents hydrographs for R-1 and observation wells MW-1S, MW-6D, MW-7D, MW-8S, MW-8D, MW-11D, B-9 and FW-2. An immediate drawdown of about fifty feet is observed in R-1 at the start of pumping. Immediate water level responses are observed at MW-6D, MW-7D, B-9 and MW-11D with declines of about three, four, seven and twenty feet at each of these wells, respectfully. More subtle water level responses are noted at MW-8S, MW-8D and FW-2. An apparent water level response is observed at MW-1S although it cannot be determined if this is in response to the pumping of R-1, R-2 or both wells. An overall maximum regional water level decline of approximately fifteen feet is apparent in all of the wells reflecting the drop in water levels due to below average recharge and the pumping of the recovery wells. The overall decline in water levels in this area is not as steep as observed in the vicinity of Mapleton No. 1 and Orton-23. This may reflect the proximity to abundant bedrock recharge in this region.

11.1.2.4 R-2

Figure 11-9 presents hydrographs for R-2 and observation wells MW-1S and MW-1D. An immediate drawdown of approximately 120 feet is observed in R-2, reflecting the low transmissivity of the materials in which this well is screened. Slight water level responses are observed at both MW-1S and MW-1D; however, it cannot be determined if this reflects the influence of R-1 or R-2 or both wells. Given the low transmissivity of the materials in which R-2 is set and the low pumping rate of about 40 gpm, the area of measurable influence of this well is limited. A five to ten foot decline in water levels is apparent in all wells over the period of record reflecting both lower recharge and the influence of pumping.

11.1.2.5 R-3

Figure 11-10 presents hydrographs for R-3 and observation wells Whiting and Booth. A drawdown of approximately 105 feet is observed in R-3, reflecting the low transmissivity of the materials in which this well is screened. Water levels in Whiting drop about twenty feet in response to pumping of R-3. A slight water level response is observed at the Booth well. Although not shown in Figure 11-10, R-3 does not have an apparent effect on water levels at MW-8S and MW-8D. The noise in the R-3 hydrograph from February to November 2000 reflects both variations in pumping rates necessary to maintain pumping water levels above the well intake and numerous starts and stops that are apparently related to power outages, often times triggered by lightning strikes. The R-3 well was shutdown between November 2000 and September 2001 in order to assess the pumping influence of the Orton-23 well and because the R-3 pump was damaged. Water levels in these three wells declined approximately ten feet over this time period. Water level declines of approximately forty-five and twenty-five feet at R-3 and Whiting were observed once the R-3 was placed into service in September 2001. An overall



Figure 11-6: Mapleton No. 1 and Selected Observation Well Hydrographs

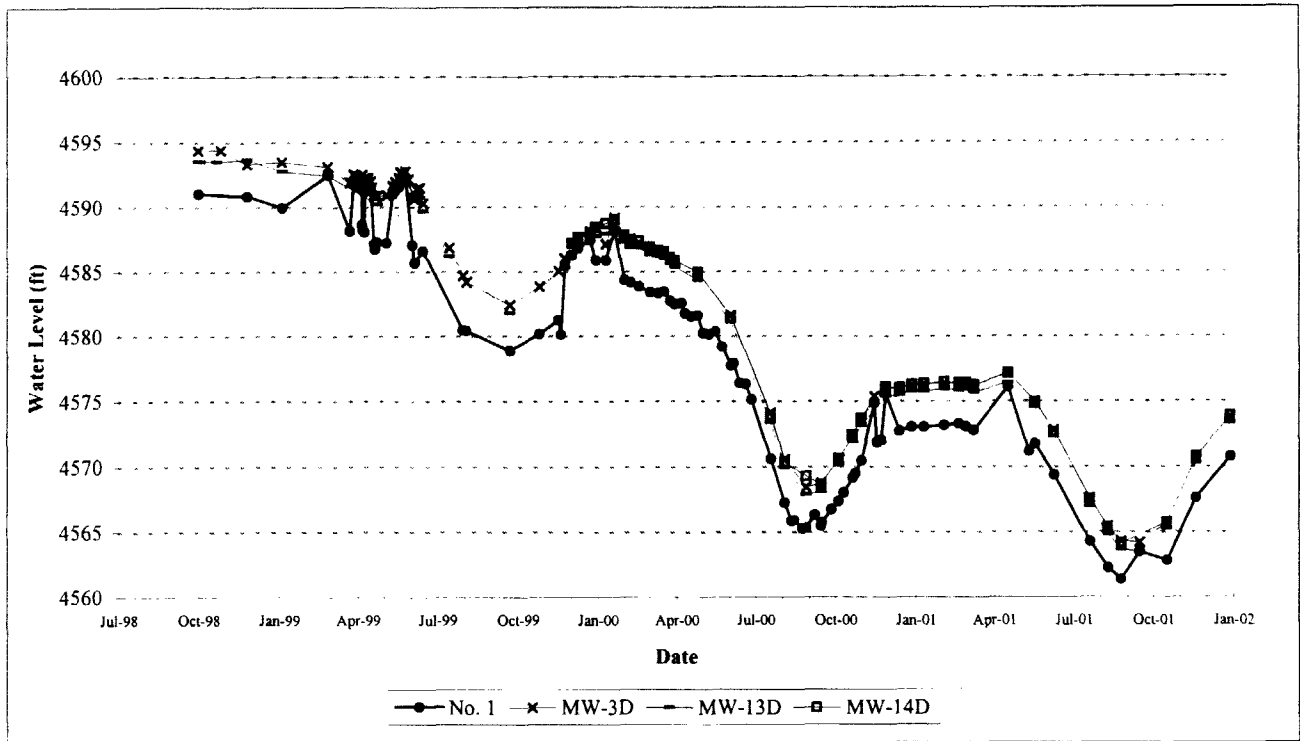


Figure 11-7: Orton-23 and Selected Observation Well Hydrographs

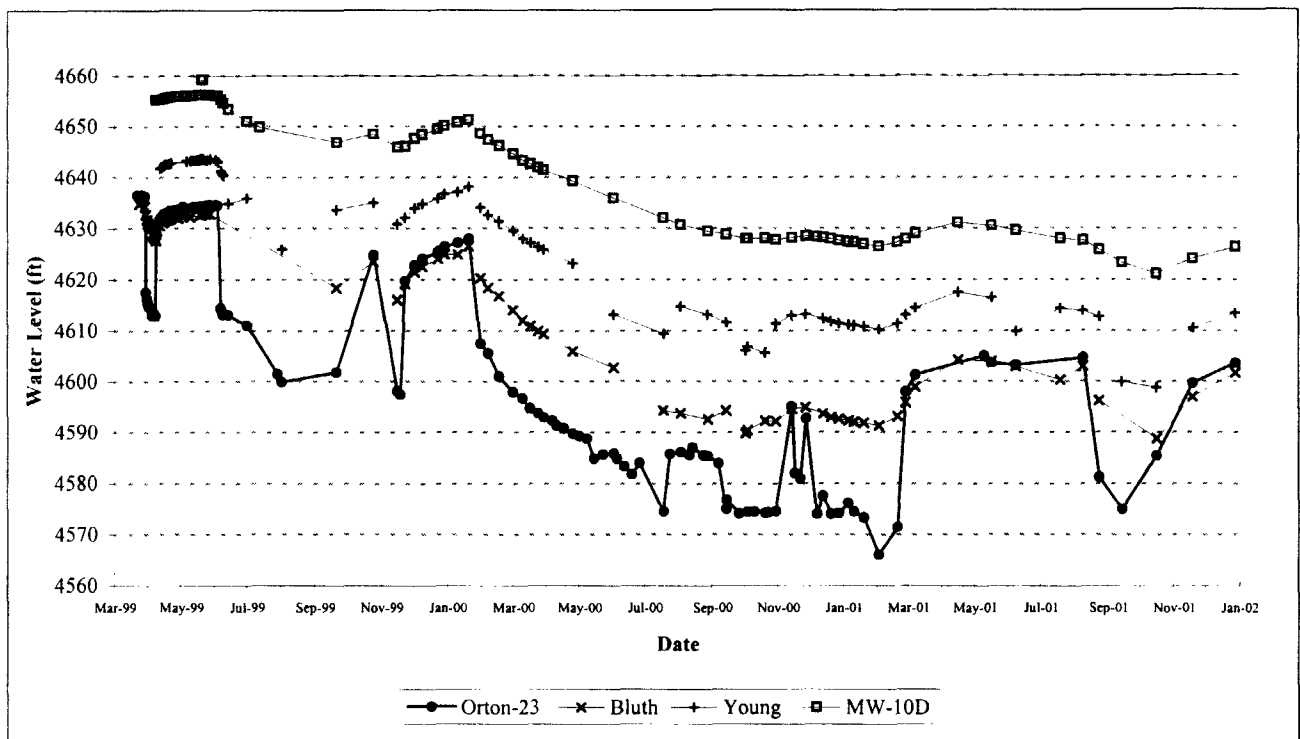


Figure 11-8: R-1 and Selected Observation Well Hydrographs

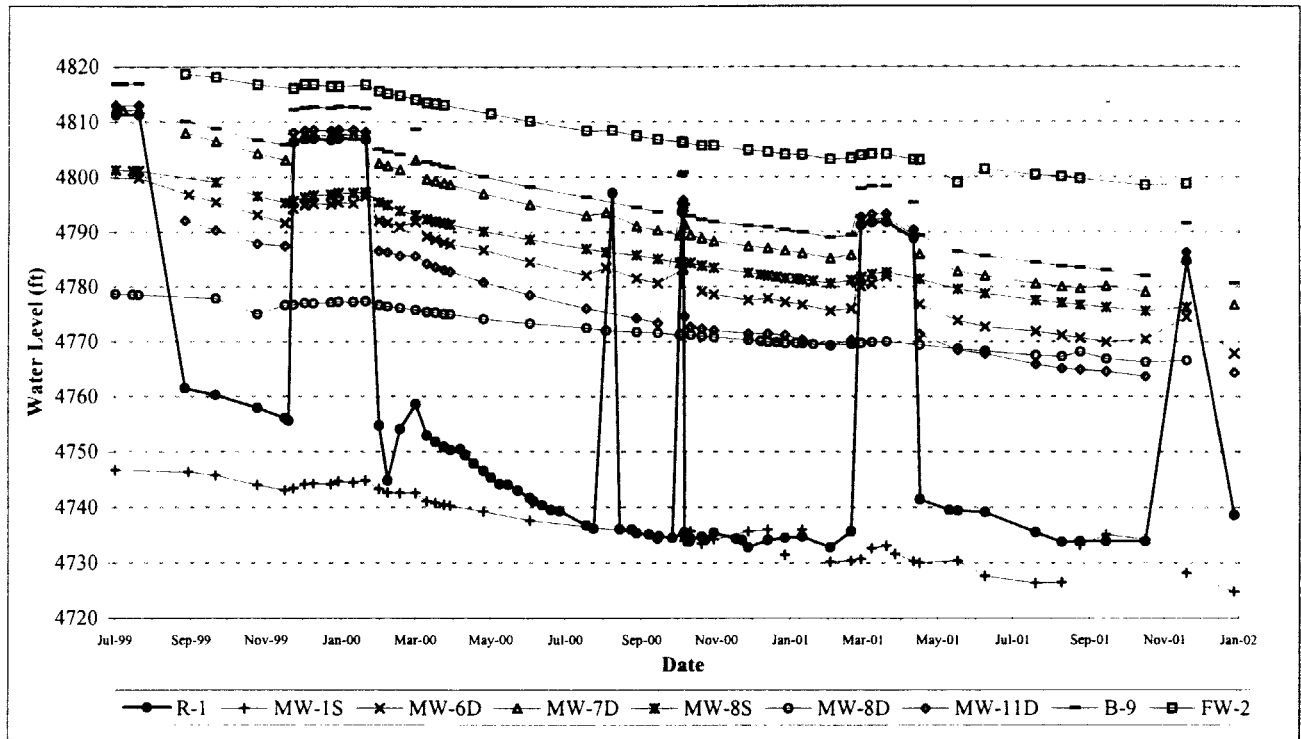


Figure 11-9: R-2 and Selected Observation Well Hydrographs

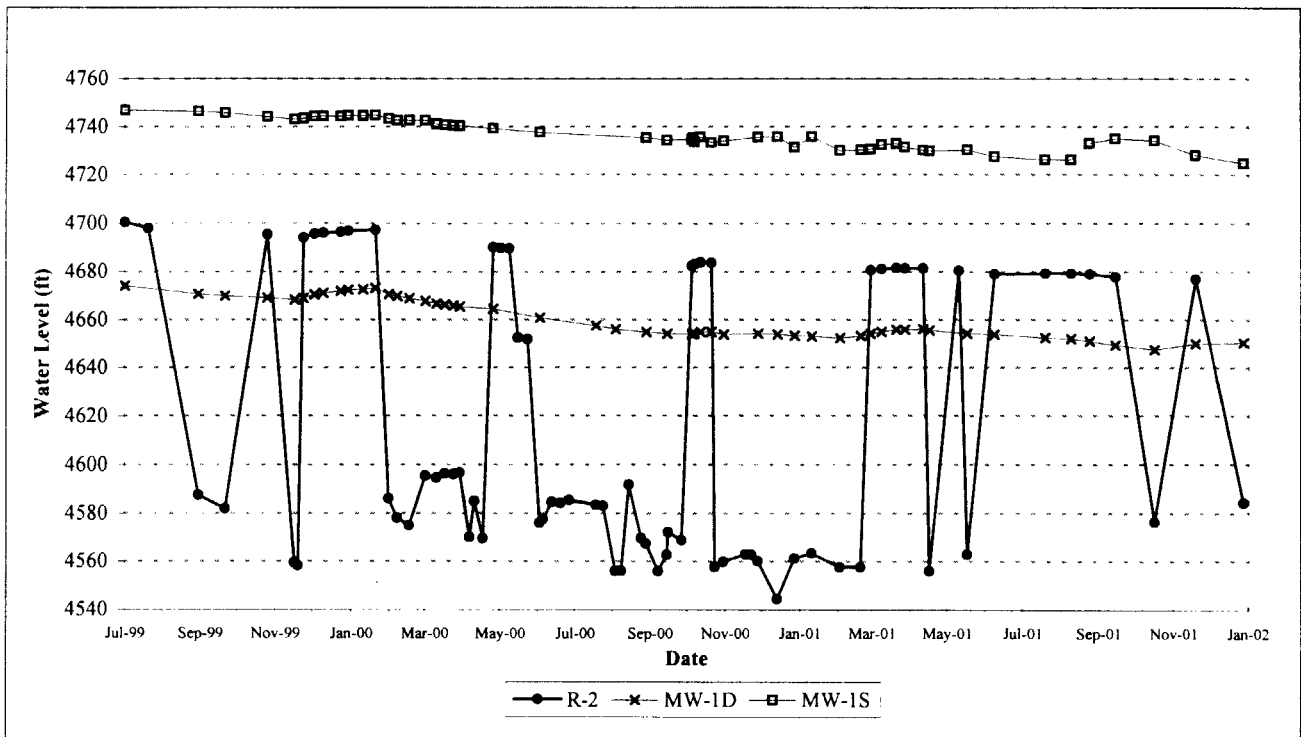
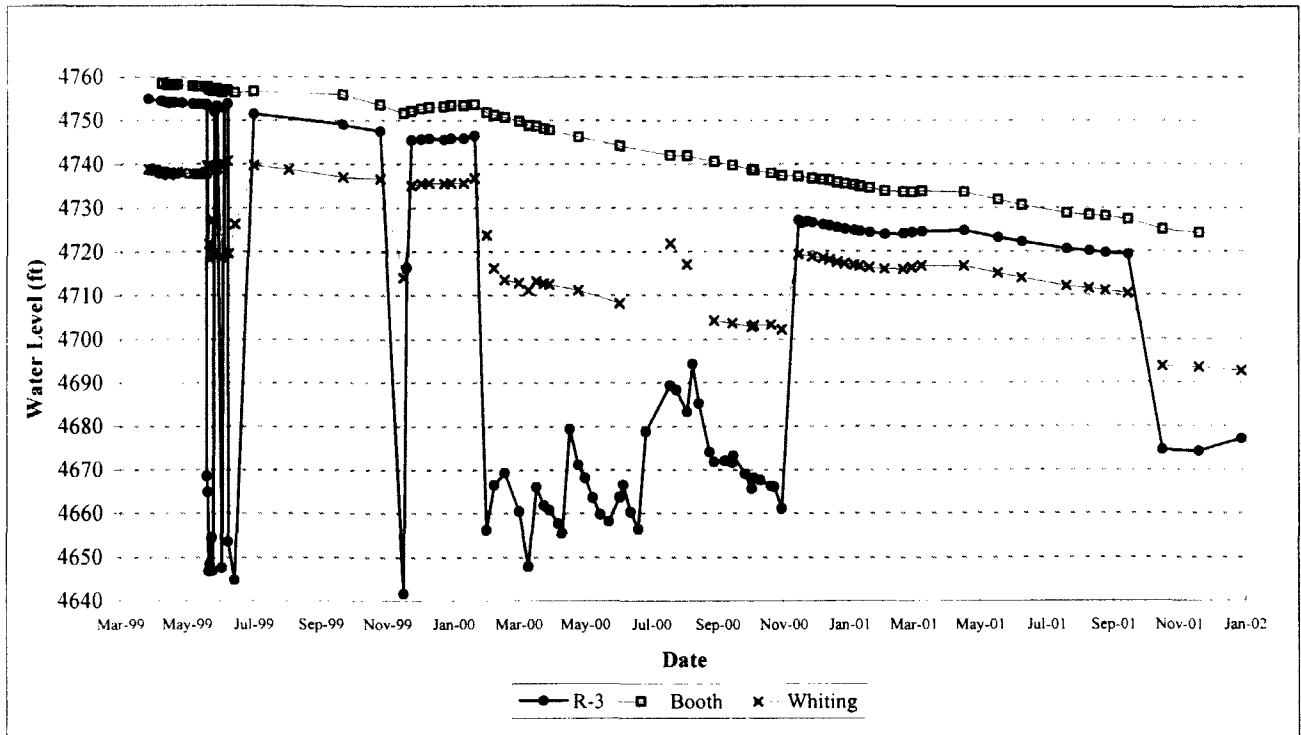


Figure 11-10: R-3 and Selected Observation Well Hydrographs



) twenty to thirty-foot decline in water levels in these wells reflects both lower recharge and the influence of pumping.

11.1.3 Approximate Containment Area

In an aquifer with a sloping water table, the capture zone of a pumping well will be smaller than the area of influence of the pumping well. This is one reason why the water level responses observed in the hydrographs cannot be used to estimate capture zones. In many cases, capture zones are estimated using numerical modeling codes such as Flowpath or analytic element models ranging from WhAEM which requires relatively simple aquifer and boundary conditions to GFlow, which can accommodate some aquifer heterogeneity and variable boundary conditions. Based on the present conceptual model of the hydrogeologic system in the study area, the numerical ground water flow model presented in the Phase IV Hydrogeologic Investigation Report (Owens Western, 1995a) does not provide an accurate simulation of the hydrogeologic system and can not be used to model capture zones. EBCo has not continued further development of a ground water model for the study area for two primary reasons: 1) to focus resources on the installation of the ground water recovery and treatment system to restore the beneficial use of the ground water resource; and, 2) concurrence by DWQ that further modeling was unnecessary.

) In the absence of a suitable numerical model, analytical models were used to approximate the steady state capture zones for the recovery wells.

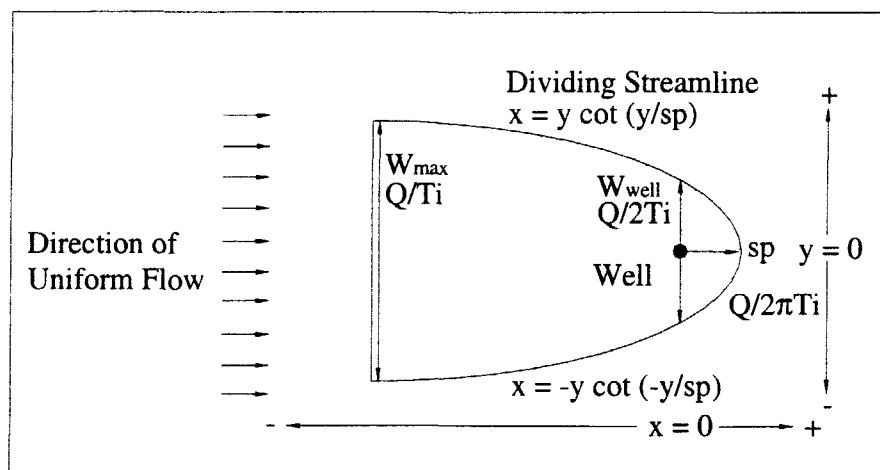
The analytical model employed assumes:

- Homogeneous and isotropic aquifer
- Uniform flow field
- Confined conditions
- Fully penetrating well
- No recharge
- No vertical flow
- Constant transmissivity

) Aquifer conditions at the recovery well locations may not satisfy all these assumptions. Nevertheless, analytical solutions provide a means of approximating capture zones for the recovery wells. Figure 11-11 presents the equations used to calculate the steady state capture zone for a single well in a homogeneous, isotropic, confined aquifer with a uniform regional hydraulic gradient.



Figure 11-11: Analytical Solution for the Capture Zone of a Single Well



The variables are defines as follows:

- Q = Discharge Rate (ft³/day)
- T = Transmissivity (ft²/day)
- i = Hydraulic gradient (unitless)
- sp = Stagnation point (ft)
- W_{max} = Maximum width of the capture zone infinitively far upgradient of the well (ft)
- W_{well} = Width of the capture zone at the pumping well (ft)

The equation for the dividing streamline is solved for several values of y. The dividing streamlines, maximum upgradient capture zone width (W_{max}), maximum capture zone width at the well (W_{well}) and the stagnation point (sp) were calculated for each recovery well. Table 11-1 lists the values of the parameters used to calculate the approximate steady state capture zones for each recovery well, as well as estimated W_{max}, W_{well} and sp values.

Table 11-1: Values Used For Estimation of Recovery Well Capture Zones

Well	Q (gpm)	Q (ft ³ /d)	T (ft ² /d)	i	W _{max} (ft)	W _{well} (ft)	Sp (ft)
Mapleton No. 1	1050	202139	71000	0.0006	4745	2373	755
Orton-23	850	163636	9660	0.01	1694	847	270
R-1	350	67380	2930	0.01	2300	1150	366
R-2	35	6738	230	0.07	419	209	67
R-3	50	11551	240	0.07	688	344	109

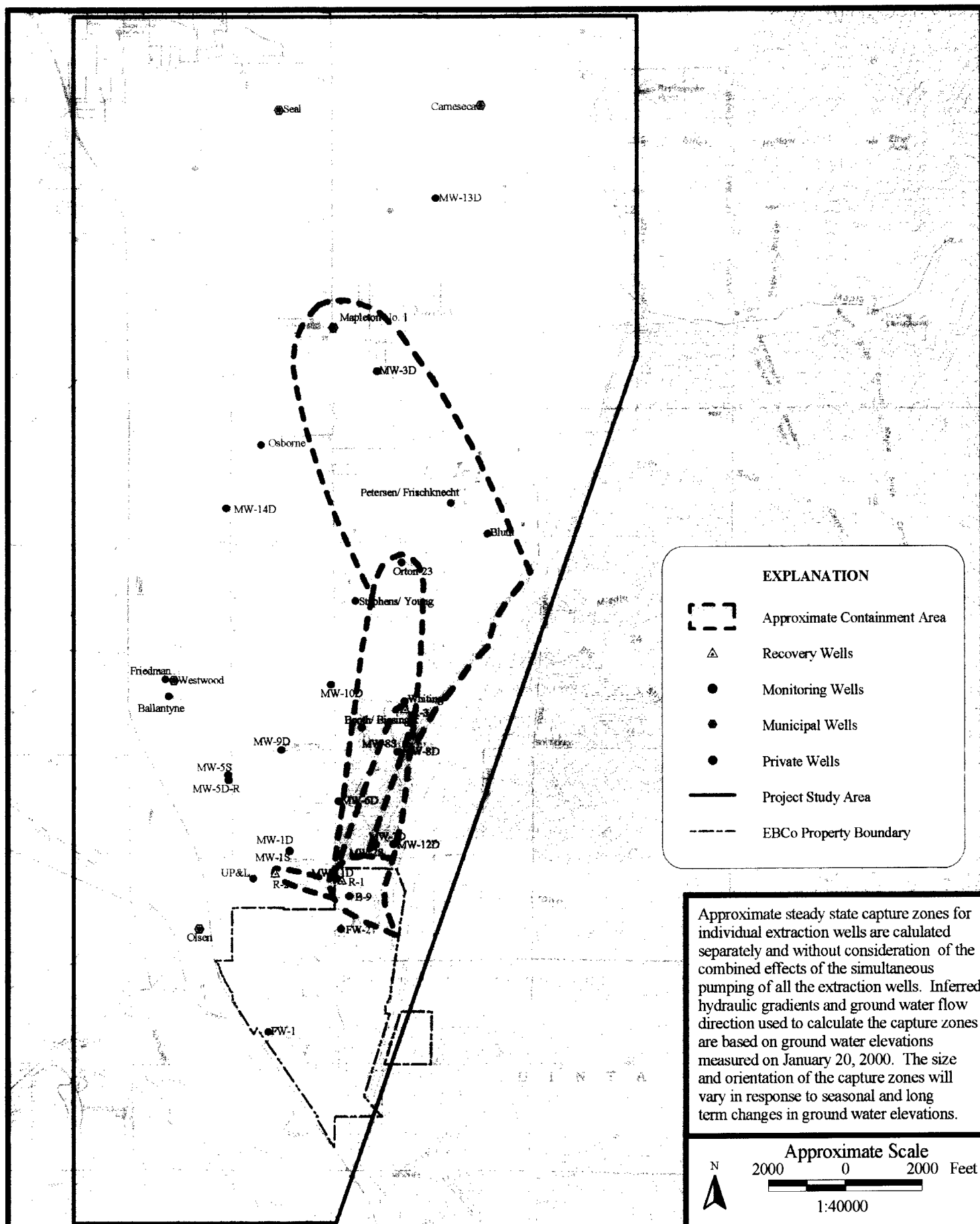


The approximate steady state capture zones calculated for the five extraction wells are illustrated in Figure 11-12. The orientation of the capture zones reflects inferred ground water flow directions based on the approximate water table elevation contours for January 2000. This approximation does not account for variations in transmissivity, hydraulic gradient, recharge and anisotropy that are inherent to the regional aquifer, nor are the combined effects of pumping from all the extraction wells considered. In actuality, the capture zones are likely to have more variable shapes reflecting the heterogeneity of the regional aquifer. Nevertheless, this is a useful tool for illustrating an approximate containment area boundary.

The approximate containment area established by the pumping wells indicates the following:

- The steady state capture zone for Mapleton No. 1 is relatively large. The width of the containment area extends from the vicinity of the Bluth and Frischknecht wells in the east to the area of the Evans and Young wells in the west. It appears that solutes within this area will flow toward the Mapleton No. 1 well, and that pumping of the Mapleton No. 1 well will prevent the possible movement of solutes in this region to the north and northwest. Monitoring will be used to confirm this capture zone.
- It appears that pumping of the Orton-23 well will capture solutes moving northward along the mountain front where preferential northerly ground water flow pathways are inferred to be present in the regional aquifer. Assuming this condition, pumping of Orton-23 should cut off the northward migration of solutes in this area. Based on the available data, it is uncertain if pumping of Orton-23 and Mapleton No. 1 will capture solutes present at the locations of MW-10D and Young. This will be the subject of ongoing evaluation during the data collection program.
- The steady state capture zone estimated for R-3 appears to fall entirely within the steady state capture zone estimated for Orton-23. Assuming this is true, solutes migrating through the regional aquifer in the vicinity of R-3 will eventually be captured by Orton-23. This suggests that the pumping of R-3 is not necessary for the maintenance of the containment area. However, R-3 is open to relatively low permeability deposits and pumping and monitoring of this well may provide useful data about the behavior of solutes in fine-grained aquifer materials. Also, the periodic starting and stopping of this well to assess solute behavior as part of the monitoring plan should not affect the containment area.
- Pumping of R-1 appears to capture solutes that may be present in the regional aquifer below historic source areas in the northeast corner of the Plant. It appears that pumping of R-1 will cut off the north and northwesterly migration of solutes from this area. The steady state capture zone estimated for R-1 is similar to the capture zone estimated for this well using numerical modeling methods (Charter Oak, 1999). Section 6.7.3.1 of this CAP discusses the preliminary conclusion that





4505 South Wasatch Blvd., Suite 360
Salt Lake City, Utah 84124
TEL: (801) 277-6150
FAX: (801) 277-6151

Approximate Containment Area

FIGURE 11-12

the perched ground water identified in the northeast area of the Plant is not having an adverse impact on the regional unconsolidated aquifer. In addition to that discussion, it is important to note that recovery well R-1 is presently capturing regional ground water in the area that would potentially be impacted by possible very slow releases from the perched ground water system. Therefore, the current recovery well system is protective of water quality in the regional unconsolidated aquifer. If the ongoing site investigation work indicates that corrective measures are needed to address this perched ground water; recommendations will be made as necessary.

- The estimated steady state capture zone for R-2 is limited in extent. R-2 is only a minor contributor to the overall containment area. However, R-2 is open to relatively fine grained deposits and pumping and monitoring of this well may provide useful data about solute behavior.

The hydrogeologic system in the study area is dynamic. Water level data has shown that water levels in the regional aquifer vary over a substantial range in response to variations in recharge volumes and seasonal pumping stresses. Furthermore, the magnitude of the water level changes is not uniform across the study area. From 1994 through 1998 water levels in southeast portions of the study area near the northeast corner of the Plant and Crowd Canyon alluvium rose over fifty feet in some locations. Water levels in wells located to the west and north only rose about twenty feet over this same time period. Locally variable changes in water levels will affect hydraulic gradients and may affect ground water flow directions. Therefore, the size, shape and orientation of the zone of capture, both for individual wells and the combined extraction system, will vary through time. In particular, seasonal or longer term changes in ground water flow directions will change the orientation of the capture zones. These temporal changes in ground water flow direction and capture zone orientation effectively increases the size of the containment area. The approximate area of containment of the extraction system will be the subject of ongoing evaluation as part of the proposed monitoring program.

11.1.4 Volumes Extracted

Table 11-2 provides a summary of the volume of water extracted from each recovery well through the end of calendar year 2001. Nearly three billion gallons of water have been extracted and treated since the beginning of recovery system operations in August 1998. Of this total approximately three hundred and sixty million gallons was beneficially used in the Mapleton and Spanish Fork PI systems during 2000 and 2001.



Table 11-2: Total Volume of Ground Water Extracted and Treated

Extraction Well	Approximate Volume Extracted (million gallons)
R-1	451
R-2	30
R-3	47
Orton-23	662
Mapleton No. 1	1,504
Total	2,692

Table 11-3 presents estimates of annual average recovery system performance based on expected average pumping rates and average constituent concentrations from extraction wells since pumping began. For this analysis the flow rates for R-1, R-2, R-3, Orton-23 and Mapleton No. 1 are 350, 35, 50, 850 and 1,000 gpm, respectively. Nitrate, RDX and total specialty nitrate esters (a combination of EGDN, DEGDN, TEGDN, TMETN, BTTN, PETN) concentrations were used for this assessment. Orton-23 and Mapleton No.1 account for approximately 81% of the annual volume of ground water extracted. R-1 accounts for about 15% of the annual volume and R-2 and R-3 combine for the remaining 4%. Based on a percentage of approximate annual constituent mass recovery, Orton-23 recovers the highest mass percentage of nitrate (45%), RDX (55%) and total specialty nitrate esters (31%). This reflects both the relatively high flow rate and higher concentrations of constituents in the Orton-23 well. Mapleton No. 1 provides 38% of the total mass recovery of nitrate, 13% of RDX and 17% for total specialty nitrates. R-1 recovers approximately 31% of the total mass recovery of RDX, which reflects the higher concentrations of RDX in R-1. R-2 and R-3 combined account for less than 4% of the annual mass recovery of nitrate, RDX and total specialty nitrate esters.

Because the forgoing analysis is subject to bias due to the volume of water extracted, it is illuminating to look at the percentage of mass removed normalized to a fixed volume of water extracted. This type of analysis provides insight into the relative efficiency of constituent removal. Table 11-4 provides a summary of percentage of mass of solutes removed per each 100,000 gallons pumped. Recovery wells R-2 and R-3, which have the lowest flow rates and therefore the lowest percentage of solute removal on an annual volume basis, generally have solute removal efficiencies that are similar to the other higher volume wells. Conversely, the solute removal efficiency of Mapleton No. 1 drops noticeably, particularly for CEMs, when compared to the other recovery wells, reflecting the lower concentrations of CEMs in Mapleton No. 1.

When evaluating these data, Tables 11-3 and 11-4 should be considered together. Considering both flow rate and removal efficiency, R-1 and Orton-23 are the best performing wells for solute recovery and R-2 and R-3 exhibit the lowest solute recovery performance.

The projected annual discharge of the recovery system of approximately 3,686 acre-feet per year is compared with recharge estimates for the regional aquifer within the study



Table 11-3: Annual Average Recovery System Performance

Well	Total Volume Recovered (gal)	Relative Volume Percentage	Ave NO ₃ -N (mg/L)	Total NO ₃ -N (lb)	Percentage NO ₃ -N Recovery	Ave RDX (mg/L)	Total RDX (lb)	Percentage RDX Recovery	Ave TSNE (mg/L)	Total TSNE (lb)	Percentage TSNE Recovery
R-1	183,960,000	15%	6.78	10,386	14%	18.47	28	31%	6.78	10	11%
R-2	18,396,000	2%	10.33	1,582	2%	ND	0	0%	6.64	1	1%
R-3	26,280,000	2%	4.36	954	1%	12.34	3	3%	6.53	1	2%
Orton-23	446,760,000	37%	8.95	33,296	45%	13.02	48	53%	7.56	28	31%
No. 1	525,600,000	44%	6.47	28,317	38%	2.73	12	13%	3.62	16	17%
TOTALS	1,200,996,000	100%		74,535	100%		91	100%		57	100%

ND = Not Detected

**Table 11-4
Recovery Well Solute Removal Efficiency
Mass Removed Per 100,000 Gallons Extracted**

Well	Total NO ₃ -N (lb)	Relative Percentage NO ₃ -N	Total RDX (lb)	Relative Percentage RDX	Total TNSE (lb)	Relative Percentage TNSE
R-1	5.6	18%	0.015	40%	0.006	22%
R-2	8.6	28%	0.000	0%	0.006	21%
R-3	3.6	12%	0.010	27%	0.005	21%
Orton-23	7.5	24%	0.011	28%	0.006	24%
No. 1	5.4	18%	0.002	6%	0.003	12%
Totals	30.7	100%	0.039	100%	0.026	100%

area. Assuming that the 1990 recharge estimated by Brooks and Stolp (1995) is approximately 65% of the average annual recharge, the average annual recharge to the regional aquifer from all sources is about 29,466 acre feet per year. The projected annual recovery system discharge is approximately 13% of the total average annual recharge to the study area. Because much of the total study area recharge is outside of the zone of capture of the recovery wells, comparison of recovery system discharge to recharge occurring along the eastern margin of the study area in the vicinity of the recovery wells is probably more relevant. The average annual recharge along the eastern boundary of the study area, from the northeast corner of the Plant to Maple Canyon, is approximately 11,960 acre-feet per year. The projected annual recovery system discharge is approximately 31% of this total. While these figures do not speak directly to the adequacy of the ground water extraction system, they do lend a perspective on the magnitude of the ground water extraction.

11.1.5 Constituent Concentration Trends

Operation of the ground water recovery system has allowed the collection and preliminary evaluation of water quality data in the context of recovery system performance monitoring. The available data record is insufficient to make any determinations or predictions about future solute behavior and recovery system performance, at this time. This is especially true in the complex hydrogeologic setting of the study area. It is expected that several additional years of data collection will be required before the ground water solute concentration trends will provide meaningful insight into future solute behavior under the effects of pumping.

Concentration trend charts for each of the recovery wells and selected observation wells are provided and discussed below. Water quality data from the third quarter of 1995 through the fourth quarter of 2001 have been included in this analysis. The third quarter of 1995 was selected as a starting point because that is when consistent sampling for CEMs began and it offers sufficient pre-pumping data to assess if pumping has affected apparent changes in solute concentration trends in selected observation wells. Charts have been prepared for nitrate-nitrogen, RDX and total specialty nitrate esters (TSNE). Charts for HMX were not prepared due to its relatively limited distribution and low concentrations. The vertical dashed line in each chart indicates the approximate start of pumping for each recovery well. The statistical significance of the concentration trends was evaluated using the Mann-Kendall Trend Test as described previously in this document.

11.1.5.1 Mapleton No. 1

Figure 11-13 presents concentration trend charts for Mapleton No. 1 and MW-3D.

Nitrate-nitrogen concentrations in Mapleton No. 1 have no trend. Nitrate-nitrogen concentrations in Mapleton No. 1 exhibit possible seasonal variations after the commencement of pumping as compared to relatively constant concentrations prior to the



start of pumping. These data were not corrected for apparent seasonal variation and that may affect the statistical results. There is no trend in nitrate-nitrogen concentrations at MW-3D.

No trend in RDX concentrations is observed in Mapleton No. 1. Possible seasonal variations in RDX concentration are not as obvious as those observed for nitrate-nitrogen. An upward RDX concentration trend is present in MW-3D. This may reflect the movement of RDX from higher concentration areas located to the southeast of MW-3D under the influence of pumping of Mapleton No. 1. RDX concentrations in MW-3D are approximately 2 µg/L higher than they were prior to the start of pumping.

TSNE data exhibit wide variability in concentrations in both Mapleton No. 1 and MW-3D, particularly after pumping began. An apparent spike in concentrations was observed immediately after pumping began, dropping to a low in October 1999 and rising to a more or less consistent concentration level through 2000. TSNE concentrations in both Mapleton No. 1 and MW-3D were generally lower in 2001 than in 2000. The apparent variability in TSNE concentrations may be due in part to variations in low concentrations of these compounds as a function of the analytical method. The combining of these compounds into a single value serves to amplify the concentration variations in the individual compounds.

11.1.5.2 Orton-23

Figure 11-14 presents concentration trend charts for Orton-23, MW-10D, Young and Bluth.

Nitrate-nitrogen concentrations in Orton-23 have a downward trend, although it is not readily apparent if the decreasing trend is in response to pumping. The apparent seasonal variations in nitrate-nitrogen concentrations observed in Mapleton No. 1 are not observed in Orton-23. An increasing trend in nitrate-nitrogen concentrations is observed at MW-10D, and a downward trend is observed at Young. No trend in nitrate-nitrogen concentrations is observed at Bluth.

RDX concentrations in Orton-23 have no trend. This contrasts with the upward trend that was calculated using data through the year 2000. This may be due, in part, to the reduced pumping in the Orton-23 well from March through August of 2001. Increasing RDX concentration trends are observed in both MW-10D and Young, although RDX concentrations appear to have been increasing at these locations prior to pumping. No change in the slope of the trend is observed in either well after pumping began. There is no trend in RDX concentrations at Bluth.

As observed in Mapleton No. 1 and MW-3D, TSNE data exhibit wide variability in concentrations in these wells. No trends in TSNE concentrations are present in Orton-23, MW-10D, Young or Bluth.



11.1.5.3 R-1

Figure 11-15 presents concentration trend charts for R-1, MW-6D, MW-7D and MW-11D.

Downward trends in nitrate-nitrogen concentrations are present in R-1, MW-6D, MW-7D and MW-11D. Long-term declining concentrations in MW-6D, MW-7D and MW-11D appear to have leveled off. Based on the available data, the declining nitrate-nitrogen trends in these wells do not appear to have been affected by the pumping of R-1.

Downward trends in RDX concentrations are present in R-1, MW-6D, MW-7D and MW-11D. Long-term declining trends are apparent in each of the observation wells. Pumping does not appear to have influenced RDX concentration trends in MW-6D or MW-7D. However, at MW-11D an apparent acceleration in the rate of decline in RDX concentrations is observed after pumping began.

No trend in TSNE concentrations is present in R-1. Long-term, declining TSNE concentrations trends are observed for each of the selected observation wells. It is not clear from the available data if pumping of R-1 has affected TSNE concentrations in the observation wells.

11.1.5.4 R-2

Figure 11-16 presents concentration trend charts for R-2, MW-1S, MW-1D and UP&L.

Nitrate-nitrogen concentrations in R-2 have remained stable at about 10 to 11 mg/L since pumping began with no trend. Downward trends in nitrate-nitrogen concentrations are present in MW-1S, MW-1D and UP&L. Since pumping began, nitrate-nitrogen concentrations in the selected observation wells appear to have remained relatively stable at concentrations of about 6 to 8 mg/L.

RDX has not been detected in R-2. No trends are present for RDX in MW-1S, MW-1D or UP&L.

EGDN is the only specialty nitrate detected in R-2. The concentration of EGDN in R-2 has been relatively steady at about 6 µg/L with no trend. Declining trends in TSNE concentrations are present in MW-1S and UP&L. No trend is present at MW-1D.

As noted earlier in this document, nitrate-nitrogen and EGDN are two of the most mobile and widespread constituents detected in the regional aquifer. The reason that only these two compounds have been detected in R-2 at this time may be because the other COCs, if present, are retarded along the ground water flow path due to the fine-grained nature of the deposits in which R-2 is screened. It is also notable that nitrates and EGDN are among the earliest constituents discharged to the ground.



11.1.5.5 R-3

Figure 11-17 presents concentration trend charts for R-3 and Whiting.

Nitrate-nitrogen concentrations in R-3 exhibit a decreasing trend; however, a similar long-term decreasing trend is observed in Whiting before and after pumping. Based on the available data, the pumping of R-3 does not appear to have affected nitrate-nitrogen trends in this area over the period of pumping.

A downward trend in RDX concentrations is present in R-3, and RDX concentrations in R-3 have declined sharply from about 20 µg/L to 10 µg/L since the start of pumping. It appears that RDX concentrations in R-3 leveled off when the well was turned off between November 2000 and September 2001. Over the same time period at Whiting, no trend in RDX concentration is present. The pumping of R-3 appears to have had some affect on RDX concentrations in the immediate locality of the well.

No trend in TNSE concentrations is present in R-3, whereas Whiting exhibits a downward trend. Pumping of R-3 does not appear to have affected TSNE trends at this time.

The apparent declining trend observed in RDX concentrations in R-3 contrasts with the lack of obvious pumping-influenced trends for nitrate-nitrogen and TSNE. The reason for this behavior is not known at this time.

11.1.6 Estimated Time to Achieve Proposed CACLs

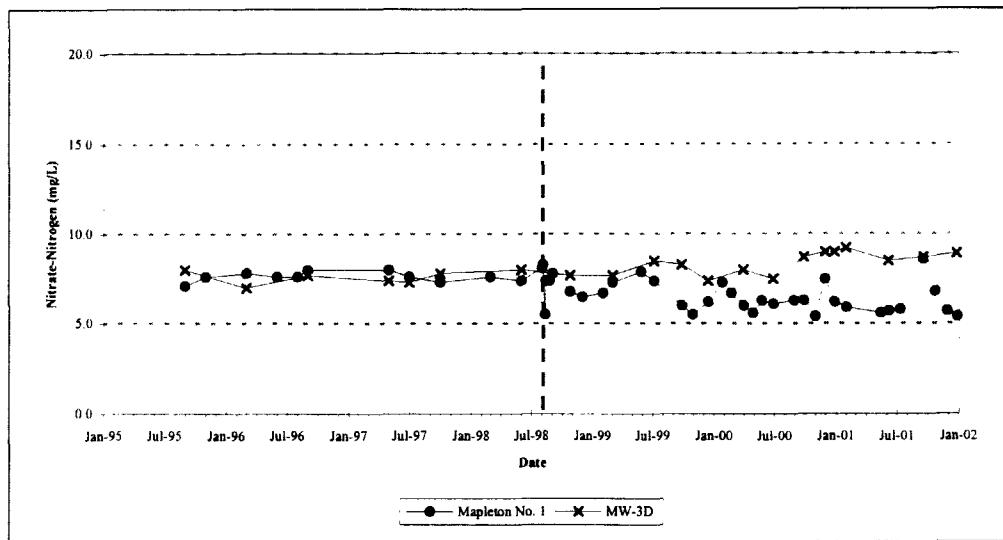
Concentrations of the specialty nitrate esters (individually or combined) are below the proposed CACL at all locations. HMX concentrations are also below the proposed CACL at all locations. Concentrations of specialty nitrate esters and HMX are not expected to exceed the proposed CACL at any time in the future.

The time to achieve proposed CACLs for the remaining COCs or the interim ground water quality goal for RDX cannot be estimated at this time for the following reasons:

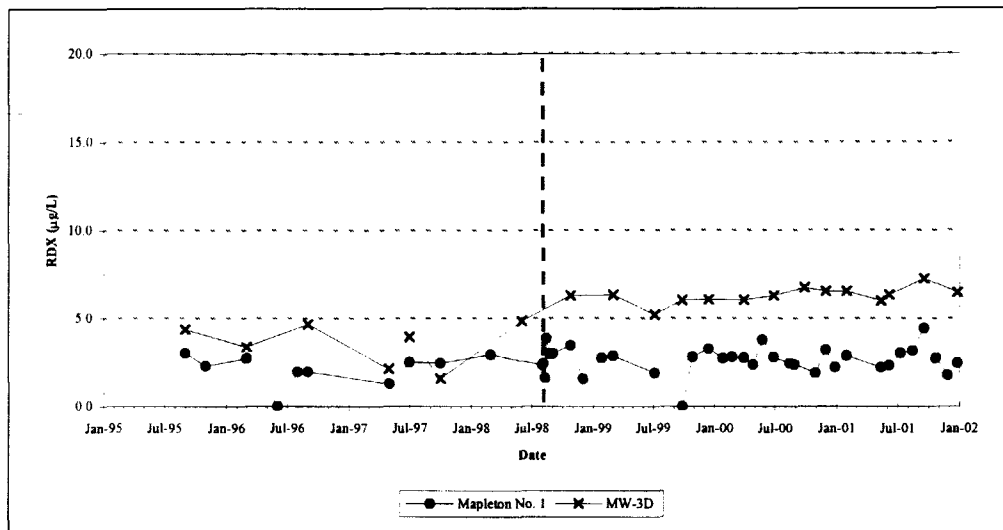
- The extraction system has not been in operation long enough to provide sufficient data for the statistical trend analyses;
- Uncertainty due to the heterogeneity of the regional unconsolidated aquifer;
- The unquantified effects of rewetting of unsaturated deposits;
- The unquantified degree to which diffusion of solutes into fine-grained deposits may act as continuing sources of solutes to the more permeable aquifer materials; and,



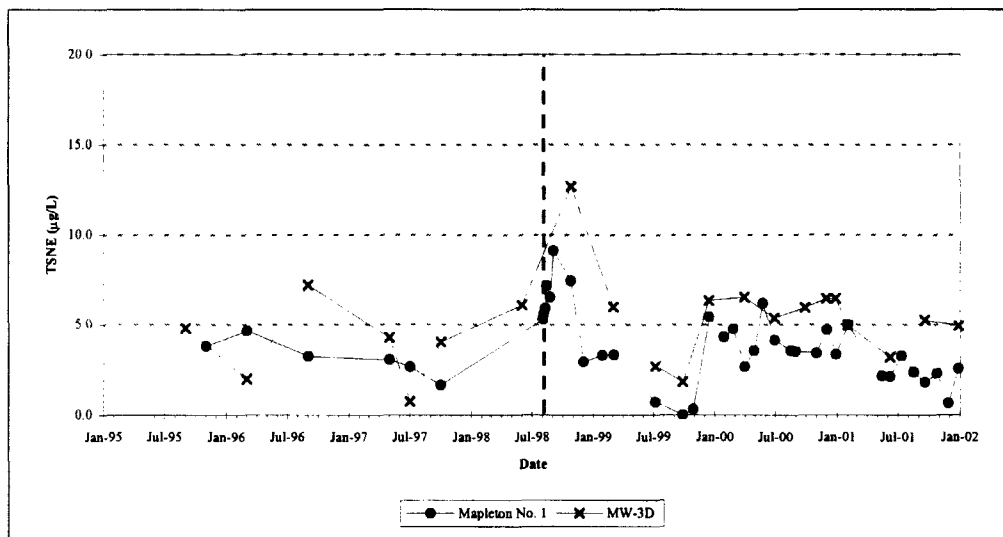
Figure 11-13: Mapleton No. 1 and Selected Observation Wells - COC Trends



Nitrate-nitrogen

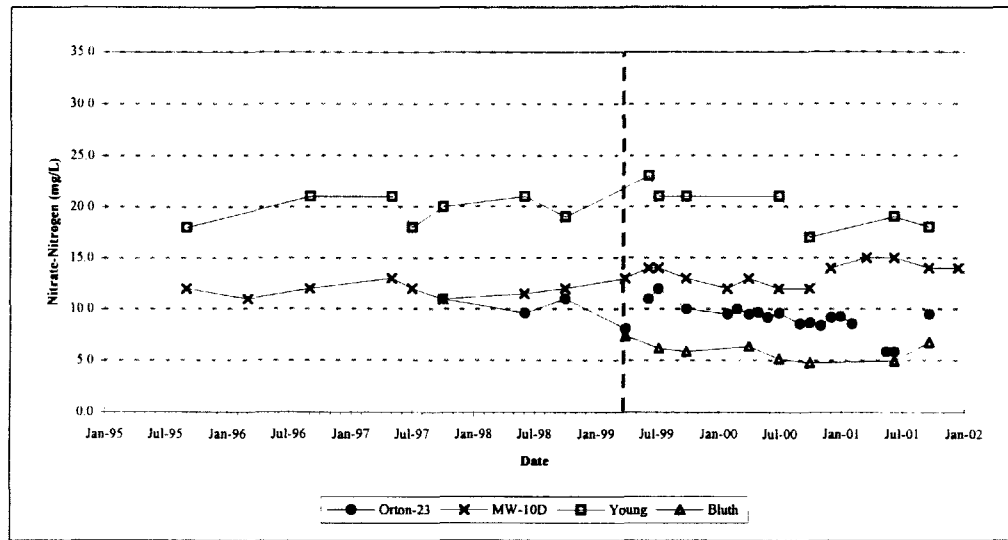


RDX

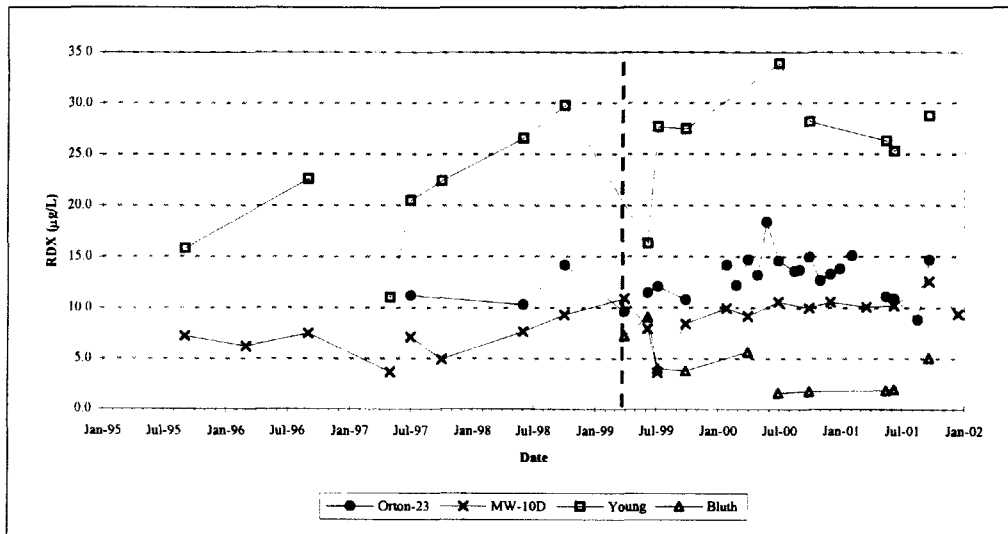


TSNE

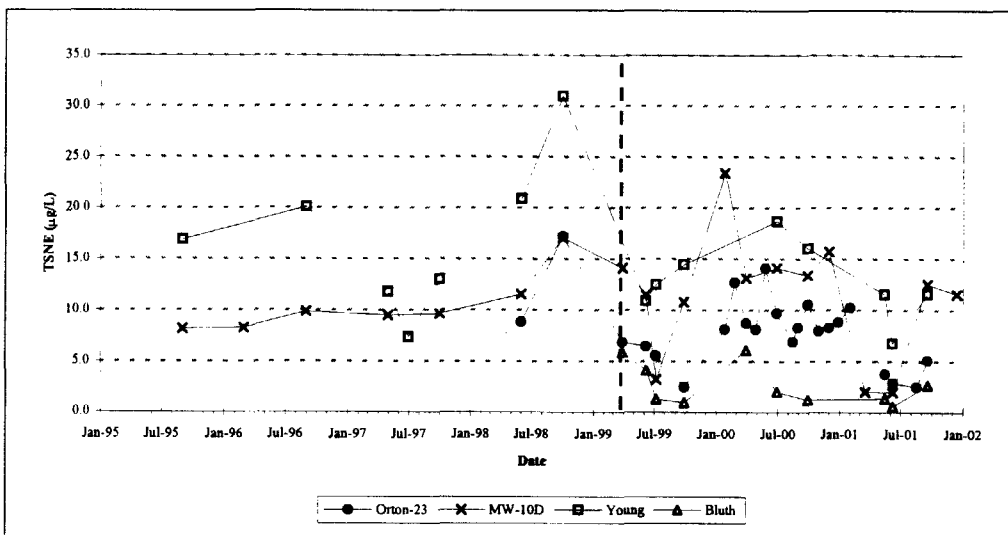
Figure 11-14: Orton-23 and Selected Observation Wells - COC Trends



Nitrate-nitrogen

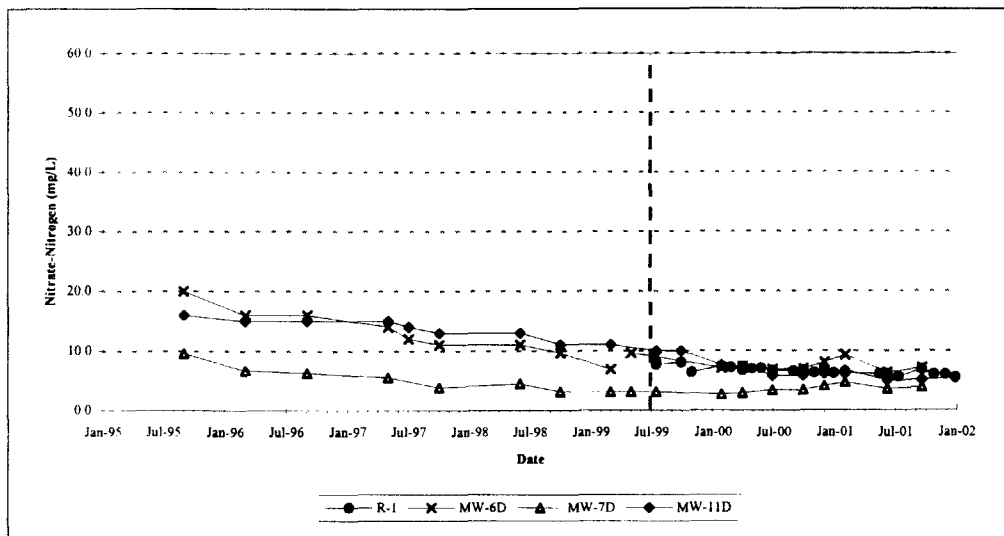


RDX

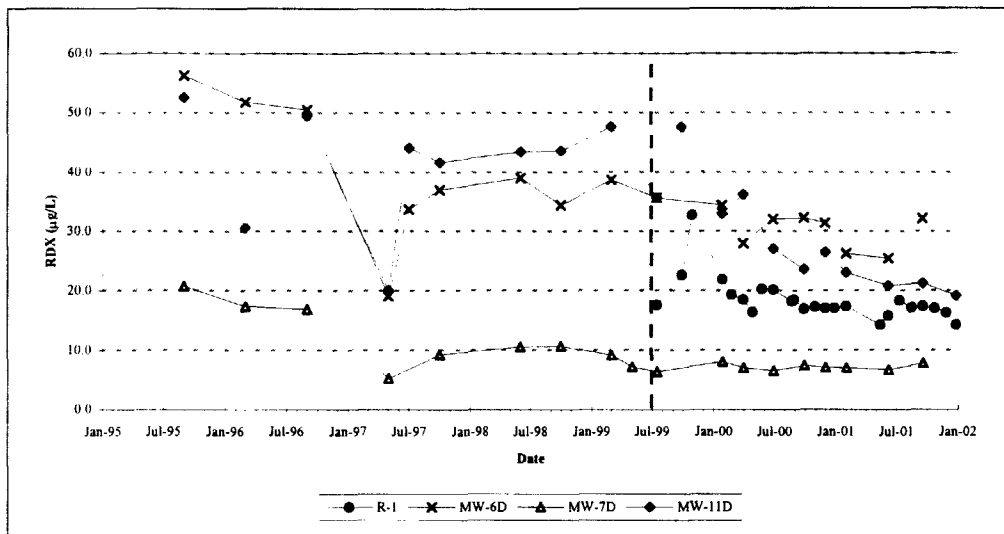


TSNE

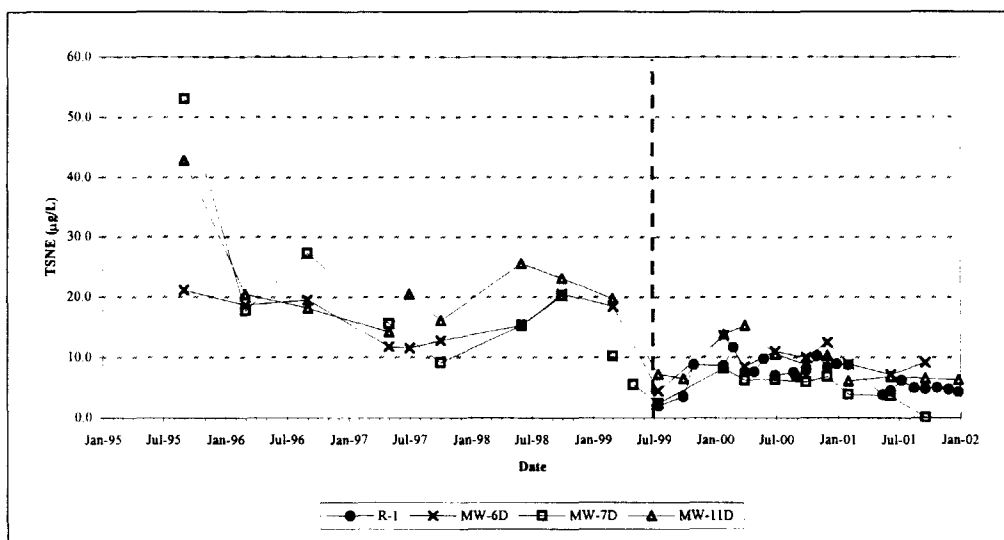
Figure 11-15: R-1 and Selected Observation Wells - COC Trends



Nitrate-nitrogen

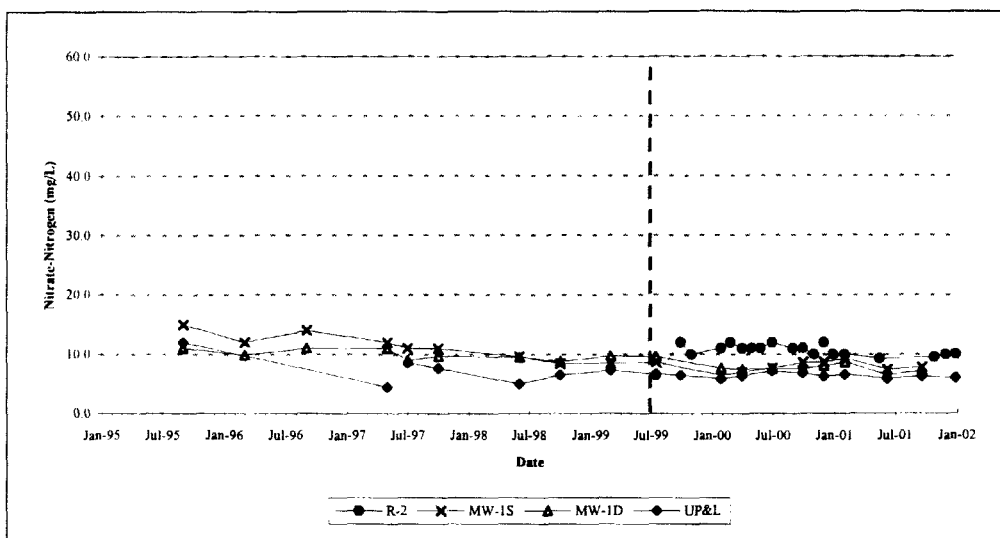


RDX

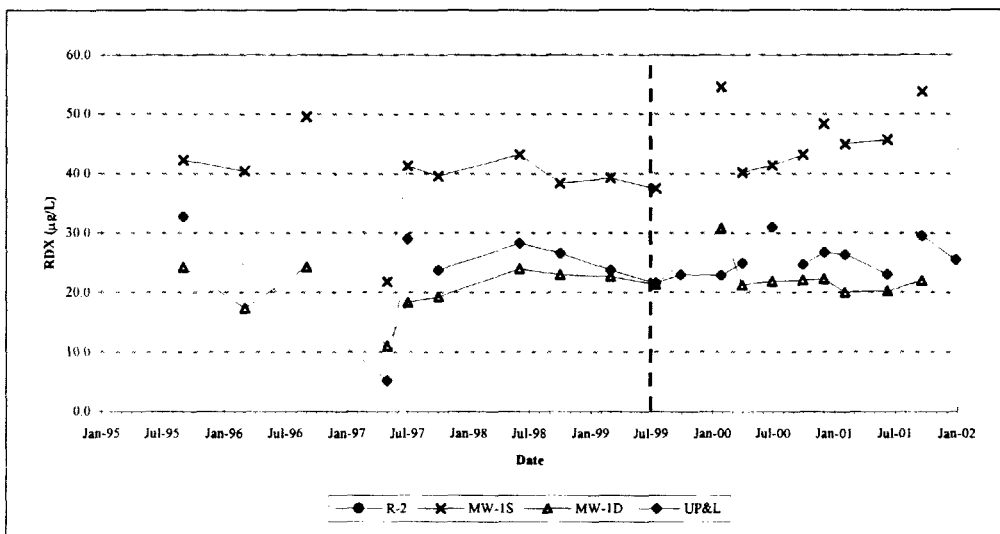


TSNE

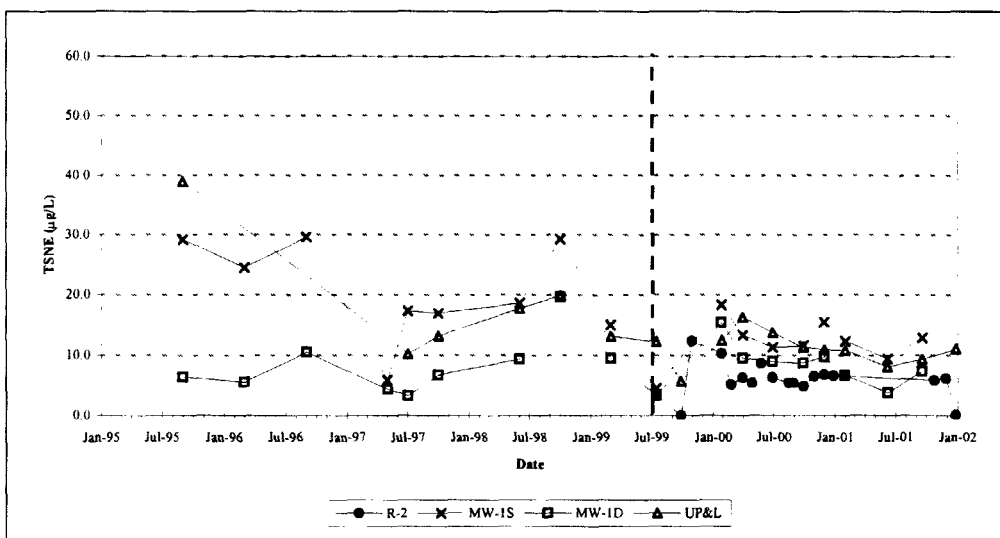
Figure 11-16: R-2 and Selected Observation Wells - COC Trends



Nitrate-nitrogen

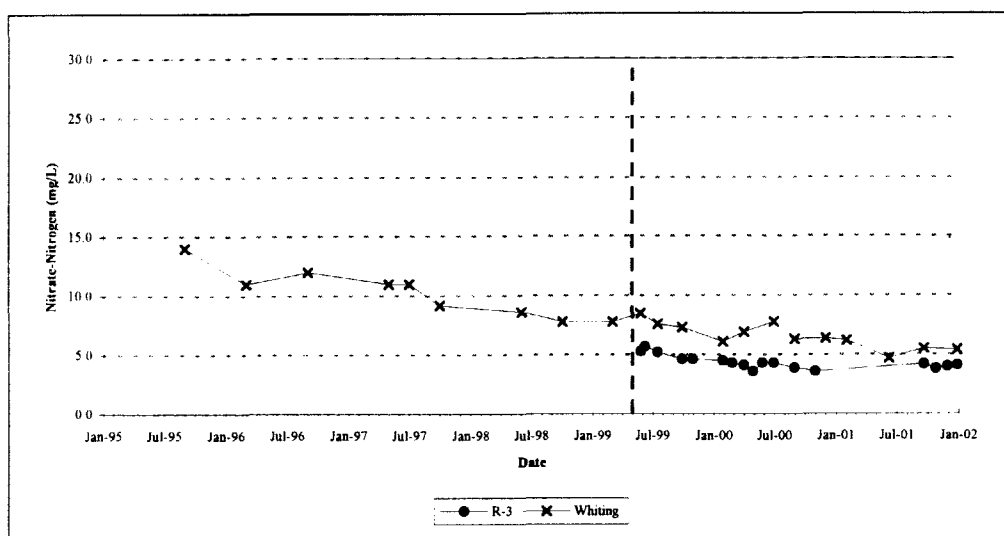


RDX

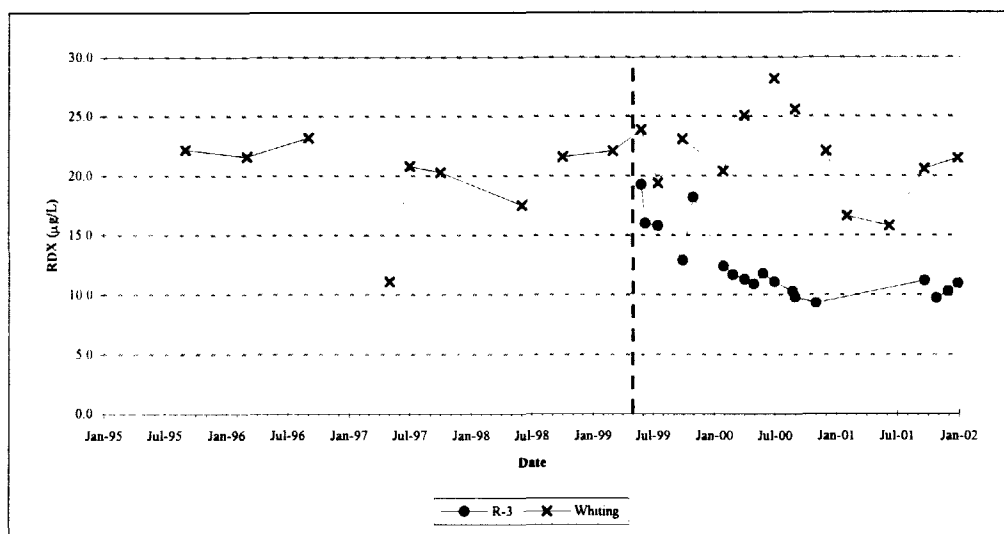


TSNE

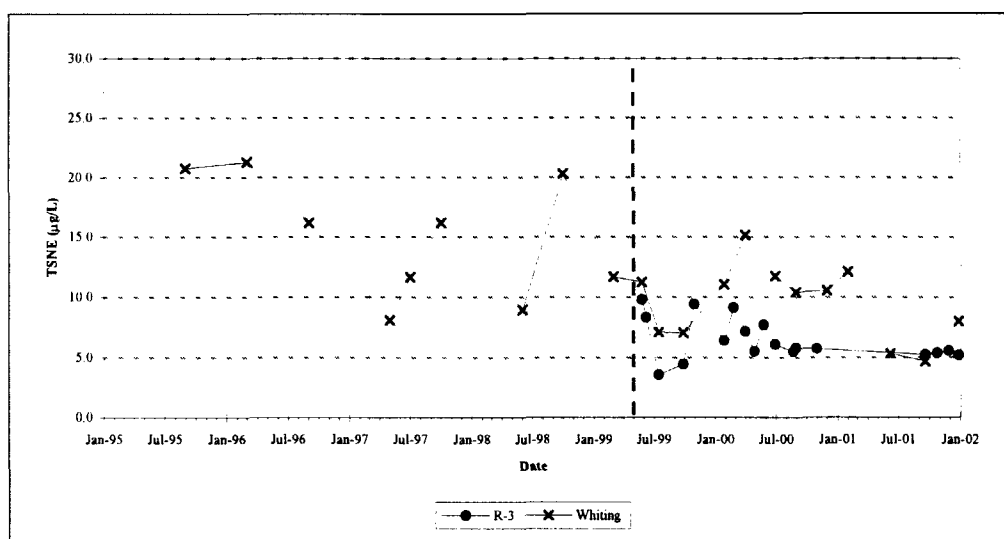
Figure 11-17: R-3 and Selected Observation Wells - COC Trends



Nitrate-nitrogen



RDX



TSNE

- The unquantified degree to which the sorption of certain CEMs will affect long-term water quality in the regional aquifer

Ongoing data collection and analysis will provide additional data that may be used to address the uncertainties identified above.

Because the total mass of COCs present in the regional aquifer is not known, a comparison of mass removed to total mass present in the regional aquifer cannot be made with any confidence. Therefore, this type of estimation tool is not suitable for this site. Likewise, due to the heterogeneity of the regional aquifer and unknown affects of diffusion and/or sorption, a simple estimation tool such as the flushing of a certain number of pore volumes of clean water is also not suitable for estimating cleanup time frames for this site.

According to "Pump-and-Treat Ground-Water Remediation – A Guide for Decision Makers and Practitioners" (EPA, 1996), unrealistic expectations for the pump-and-treat approach to site clean-up can lead to disappointment in actual system performance. The EPA document reports that the Committee on Ground Water Cleanup Alternatives of the National Academy of Sciences (NRC, 1994) has identified three major classes of sites based on hydrogeology and solute chemistry.

- *Class A.* Full cleanup to standards should be feasible using current technology. Class A sites include homogeneous single- and multiple layer aquifers involving mobile, dissolved solutes.
- *Class B.* The potential for complete cleanup to standards is likely to be uncertain. Class B sites include a wide range of hydrogeologic settings and solute chemistries that do not fall into Class A or C.
- *Class C.* Full cleanup to health-based standards is likely to be technically infeasible. Class C sites include fractured rock aquifers containing light and/or dense non-aqueous phase liquids (LNAPL and DNAPL) and single- or multiple-layered heterogeneous aquifers containing free-product DNAPL.

The classes are further divided into numerical classes 1 through 4, which indicate the relative ease of cleanup, where 1 is the easiest and 4 is the most difficult.

Table 11-5, adapted from EPA (1996) is a matrix developed by NRC (1994) identifying categories of sites for technical infeasibility determinations. The shaded boxes on the left (Class A) represent sites where cleanup to standards should be feasible with current technology. The shaded boxes on the right (Class C) represent sites where cleanup to standards is likely to be technically infeasible. The unshaded boxes (Class B) represent sites where the technical practicability of cleanup to standards is uncertain.



Table 11-5: Classes of Sites for Technical Infeasibility Determinations

Hydrogeology	Solute Chemistry					
	Mobile, Dissolved (degrades)	Mobile, Dissolved	Strongly Sorbed, Dissolved (degrades)	Strongly Sorbed, Dissolved	Separate Phase LNAPL	Separate Phase DNAPL
Homogeneous Single Layer	A (1)	A (1-2)	B (2)	B (2-3)	B (2-3)	B (3)
Homogeneous Multiple Layers	A (1)	A (1-2)	B (2)	B (2-3)	B (2-3)	B (3)
Heterogeneous Single Layer	B (2)	B (2)	B (3)	B (3)	B (3)	C (4)
Heterogeneous Multiple Layers	B (2)	B (2)	B (3)	B (3)	B (3)	C (4)
Fractured	B (3)	B (3)	B (3)	B (3)	C (4)	C (4)

The project area addressed by this CAP consists of a heterogeneous multi-layer aquifer with relatively mobile to moderately sorbed dissolved phase solutes. In accordance with the aforementioned classification the study area is considered to be a Class B(2) (bold text in table). Several additional years of data collection are required before more definitive estimates of cleanup times can be made or to determine if cleanup to proposed CACLs is technically feasible in this hydrogeologic setting.

11.2 GAC Treatment

11.2.1 GAC Treatment System Monitoring

Monthly monitoring of the GAC treatment systems as described in Section 9.1.1.2 is sufficient to assess the rate of carbon usage and the need for carbon replacement.

EGDN is generally the first CEM observed to break through the carbon treatment columns. This is consistent with the apparent low sorptive properties of EGDN relative to several of the other CEMs.

11.2.2 Carbon Exchange

Reactivation of the spent carbon is effective and appropriate for the intended use. The regenerated carbon meets the criteria established by the Utah Division of Drinking Water for the use of regenerated carbon in public water supply systems. Regenerated carbon from the three treatment systems is used interchangeably amongst the three treatment systems.



Operational experience indicates that the lead columns in both the Mapleton and Orton GAC treatment systems typically are exchanged prior to or near the start of the irrigation season that begins in April of each year. However, this schedule is continually evaluated based upon monthly carbon performance data and if the carbon in these systems exhibits a slower loading rate this annual replacement frequency may be modified. Lower flow rates through the Spanish Fork GAC treatment system results in slower carbon loading rates which means that carbon in this treatment system lasts approximately twice as long as the other two treatment systems. Ultimately, the need for carbon exchanges in all systems is based upon the results of monthly performance monitoring and compliance with effluent limits.

Data collection has shown that carbon usage is more rapid in the Orton and Mapleton GAC treatment systems relative to the Spanish Fork GAC system, even though the ground water with the highest CEM concentrations is directed to the Spanish Fork GAC system. This is due, at least in part, to the lower flow rate at the Spanish Fork GAC system (approximately 450 gpm) as compared to the Orton and Mapleton GAC systems (approximately 1,000 gpm each). However, given the lower CEM concentrations at the Orton and Mapleton systems it is unlikely that higher flow rates alone can account for the more rapid breakthrough. A thin film of food grade mineral oil has been observed in the pre-filter assemblies of the Orton and Mapleton GAC systems. Mineral oil is used to lubricate the pumps of both the Orton-23 and Mapleton No. 1 wells and some mineral oil apparently has seeped into the pump intakes or pump column. It is possible that a low (ppb) concentration of dissolved mineral oil is occupying sorption sites in the GAC and effectively lowering the CEM sorption capacity in these treatment systems. Sampling of the extracted ground water indicates that mineral oil is not present at concentrations at or above 5 mg/L; however, no analytical methods have been identified that can quantify ppb levels of mineral oil in the extracted ground water. Therefore, the potential for this phenomenon cannot be evaluated directly. Pre-filter bags that are designed to adsorb oils have been used in the Mapleton and Orton treatment systems since the spring of 2001. Although not substantially different, a comparison of carbon usage data from 2000 and 2001 suggests that there may have been a small amount of improvement in carbon loading rates with the use of the oil adsorbing pre-filters.

11.3 Natural Attenuation

Dilution and dispersion are the dominant natural attenuation mechanism in the study area. Based on available data it appears that dilution has contributed to the reduction of solute concentrations in the regional aquifer in areas of high ground water recharge. Dispersion probably has played an important role in limiting the expansion of the zone of impacted ground water around the margins of the affected area, particularly to the west. Other potential natural attenuation processes are not practically relevant in the regional aquifer. Natural attenuation, by reducing solute concentrations both within and outside of the zone of capture, supplements the installed ground water recovery and treatment system and is an integral component of the remedial action. The effectiveness of natural



attenuation at reducing constituent concentrations and limiting expansion of the affected regions will be the subject of ongoing evaluation through ground water monitoring.

11.4 Proposed CACLs

Of the COCs and provisional COCs addressed by this CAP, only nitrate-nitrogen and lead have a federal MCL or state ground water quality standard. By definition, the MCL of 10 mg/L is the CACL for nitrate-nitrogen. The Utah Ground Water Quality Standard for dissolved lead is 0.015 mg/L and is the CACL by definition. The lifetime HA for HMX is the proposed CACL for this compound. Proposed CACLs for the other constituents present in the regional aquifer have been developed based on a multi-pathway risk assessment using the latest EPA methods and exposure assumptions. The proposed CACLs consider multiple exposure pathways including drinking water, dermal contact through bathing, inhalation and the consumption of homegrown produce, fish, beef and milk. As explained in Section 9.0 of this CAP, a CACL is not proposed for RDX at this time. Rather, an interim ground water quality goal of 2 µg/L is proposed. The 2 µg/L interim ground water quality goal for RDX is numerically equivalent to the existing lifetime HA for RDX. As noted in Section 9.0, EBCo considers the HA for RDX to be founded on inadequate science and a joint DOD/EPA program is presently assessing RDX which is likely to result in future revisions to the RDX IRIS file. Until such time that the RDX IRIS file is modified, the proposed interim ground water quality goal of 2 µg/L will be used for decisions regarding extraction system operations, GAC treatment, potential human exposure and effluent limits specified in the UPDES discharges. The institutional controls that are in place serve to minimize the inadvertent use of impacted water. The proposed CACLs and the proposed interim water quality goal for RDX are designed to be protective human health with no controls on use.

11.5 Water Quality Monitoring Program

The present water quality monitoring program has been sufficient to assess water level changes and solute behavior in the regional aquifer. Weekly water level measurements from recovery wells and monthly water level measurements from selected observation wells is a suitable frequency to assess changes in water level elevations due to recovery system operation, other pumping stresses and natural seasonal variations. Water levels may be collected at a greater frequency from selected wells during recovery well start-ups and shut downs or for other reasons where the collection of additional water level data is deemed useful.

The present water quality monitoring program includes nitrate-nitrogen and the suite of CEMs identified in this CAP. A subset of wells will also be sampled for both total and dissolved lead beginning in the first quarter of 2002. Recovery wells are sampled on a monthly frequency, which is appropriate to assess the recovery system performance. Observation wells, including monitoring, private and municipal wells are sampled on a quarterly schedule.



As described in Section 12.6.2 of this document, three additional monitoring wells will be installed in 2002 to evaluate the effectiveness of natural attenuation mechanisms on the western boundary of capture. Additionally, under the auspices of the RCRA Corrective Action Program, one new on-site monitoring well open to the regional aquifer and two new monitoring wells open to perched ground water identified in the northeast area of the Plant will be constructed to assess ground water conditions in that area. The locations and design of these three RFI monitoring wells were approved by DSHW in March of 2002 and they will also be installed and sampled during 2002.

11.6 Institutional Controls

The institutional controls that are in place are protective of human health. The following activities serve to minimize the inadvertent use of impacted ground water extracted from the regional aquifer:

- An ordinance established in Mapleton City's municipal code indicates that a construction permit for a new home within City limits will not be issued without a municipal potable water connection. Mapleton City has the burden to enforce this ordinance.
- The Utah Division of Natural Resources – Division of Water Rights (DWR) has established a "restricted zone" in the study area. DWR has stated in the Ground Water Management Plan for Utah and Goshen Valley's (DWR, 1995) that applications for the transfer of ground water rights for the purposes of well construction within the restricted area will not be allowed. DWR has the burden to enforce this policy. Ground water rights in this area that are approved but unperfected may be completed as wells.
- A comprehensive review of water rights records for the study area was performed in 1997 and EBCo sent certified letters to well owners advising them of the known or suspected water quality in their well, as well as general information about the planned remediation program. Water rights records for the study area are reviewed on an annual basis to identify potential new well owners. If a new well owner is identified within the affected area, a letter will be sent to the well owner advising them of the possible water quality conditions.

In addition to aforementioned measures to minimize inadvertent use, municipal water supply wells in the study area are sampled on a quarterly schedule and the results are provided to the municipality, DWQ, DDW and the Utah County Health Department. Private wells that are part of the water quality monitoring program are also sampled quarterly and the results are provided to the well owner, DWQ and the Utah County Health Department.



11.7 On-site Controls

The degree to which on-site soils may be acting as potential sources of solutes to on-site perched ground water and the regional unconsolidated aquifer is the subject of continuing evaluation in the RFI. On-site soils that may represent a potential ongoing threat to the regional aquifer will be addressed as part of the RCRA Corrective Action at this site. Available data from the perched ground water interval identified in the northeast area of the Plant (Section 6.7.3.1 of this CAP) suggest that possible impacts to the regional aquifer from perched ground water are not significant. In addition, pumping of the R-1 recovery well is capturing the regional ground water in the area that would potentially be impacted by possible slow releases from this perched ground water system, and is therefore protective of the regional unconsolidated aquifer.

